STUDY ON THE IMPACT OF SELECTIVE LOGGING ON THE ABUNDANCE OF ANURANS IN THE RAIN FORESTS OF CENTRAL KALIMANTAN, INDONESIA

BY

AGUSTINUS WINANTO TAUFIK

A thesis submitted for the degree of Doctor of Philosophy

Institute of Ecology and Resource Management
University of Edinburgh
1997

Declaration

I hereby declare that this thesis has been composed by me and that all work presented in the thesis is my own unless specifically otherwise stated

August 4, 1997

ABSTRACT

Tropical rain forests are among the most complex natural habitats in the world and widely recognised as areas of high species diversity of both flora and fauna. Despite their biological and ecological importance tropical rain forests throughout the world are coming under pressure to be exploited for timber resources. Where management of rainforests for timber production is carried out on an unsustainable basis, serious degradation of the forest ecosystem may result, followed by loss of forest cover and/or species extinction. Therefore, there is a vital need to understand the effects of forestry practices on these diverse ecological systems and on individual species.

Vertebrates like birds and primates have been widely studied and recognised as a key indicator species for environmental stress. Anurans (amphibians) have also been identified as potentially valuable indicators of some environmental changes. The small home ranges and relatively limited dispersal ability may render amphibians useful for monitoring the effects of local environmental pertubations. In addition, their moist permeable eggs and skin also make them good bioindicators of environmental stress. Amphibians are also important in the food chain and nutrient cycling. They represent a high quality prey item (high in protein and phosphorus content) to predators. They may also play a unique role in forest nutrient cycling by regulating populations of soil invertebrates responsible for the mechanical breakdown of organic material.

Given both the important roles and functions of amphibians in forest ecosystems and the significant impacts of logging on forest ecoystems, an understanding amphibians in relation to forestry practices is required. Relatively few studies have focussed on the impact of logging on amphibians especially in the tropics. Those that have been undertaken have related mainly to temperate regions where the logging is usually clear felling and they have concentrated either on single species or on single genera. Therefore, this study will attempt to address this gap of knowledge about the impacts of selective logging on the abundance of anurans in the rain forests of Kalimantan, Indonesia. The study was undertaken from November 1994 to July 1996 at Camp 92, Central Kalimantan. Three samplingmethods were used in this study e.g. pitfall traps, searched quadrats and night riparian transects.

The results demonstrate that logging had a significant effects on the abundance of anurans and anuran species composition. Logging had a strong influence on vegetation cover, which in turn affects the temperature and humidity. Logging also affects the physical characteristic of the streams especially the bottom substrates. The combination of these associated factors strongly affects the abundance of anurans. The challenge in research for future researchers of amphibian-forestry relationships is to identifying realistic timber harvest prescriptions that best maintain those components of the forests biological legacy that are essential for healthy amphibian populations and forest ecosystem as a whole.

ACKNOWLEDGEMENTS

I wish to acknowledge my deep gratitude to my supervisors, Dr. Gareth Edwards-Jones and Dr. Juliet Vickery for their kind supervision and valuable advice given throughout the completion of this thesis. Without their constant and unequivocal support, it is doubtful that this thesis would ever reached fruition. Not at all of the people who had given of their time and energies can be mentioned here, but they may rest assured that their contributions have been appreciated.

Some persons, however, merit particular mentioned. My great debt has undoubtedly been to Coordinator of UK-Indonesia Tropical Forest Management Project - Dr. Alastair Fraser for his invaluable financial support and advice. I also would like to acknowledge the support of all people and organisation whose help and cooperation I have received.

There are various other people who, though they did not contribute to the thesis, nontheless provided invaluable and essential aid. Foremost among these are Prof. R.F. Inger, Field Museum of Natural Histrory, Chicago, USA and Dr. Boeadi, Muzeum Zoologie Bogoriense, Indonesia. To each of them, I offer my sincere thanks.

Finally, I have to acknowledge my wife Agnes, my children Adeline Carolina and Rendy Laurens, who have supported and inspired this venture and for their patience in waiting for my absence during the PhD course. Any errors are, of course, entirely my own responsibility.

Agustinus W. Taufik

LIST OF CONTENTS

LIST OF TABLES

Table

LIST OF FIGURES

Figure

CHAPTER I

INTRODUCTION

1.1. Status of Tropical Forest and Forestry Practices

1.1.1. Status of tropical forest

McNeely (1990) concluded that there exists an ongoing and unprecedented loss of variety and number of species worldwide and that this catastrophic loss in biodiversity requires immediate attention. Some biologists estimate that as much as a quarter of the total biodiversity on earth may become extinct in the next 20 to 30 years (McNeely *et al.* 1990). Rates of extinction have increased greatly in recent times due to human interference (McNeely *et al.* 1990, Wilson 1988). Habitat destruction and exploitation are among the major threats to biodiversity (Soule' 1986, Ehrlich 1988, McNeely *et al.* 1990) and this continual loss of biodiversity can have far-reaching ecological, social and economic consequences (Soule 1986, Struhsaker 1987, Wilson 1988).

Tropical rain forests are among the most complex natural habitats in the world and widely recognised as hotspots of species richness of both flora and fauna. Despite their biological importance tropical rain forests throughout the world are coming under increasing pressure to be exploited for timber resources (IUCN 1990). Where management of rain forest for timber production is carried out on an unsustainable basis, serious degradation of the ecosystem often results, followed by loss of forest cover or species extinction (Whitmore and Sayer 1992). There is a vital need to understand the effects of forestry practices on these diverse ecological systems and on the individual species. They support management strategies for the remaining areas of tropical rain forest. Only then can remaining areas be developed for the management strategies that do not reduce the biological value of these ecosystems (Howard 1986, Johns 1992).

The rate of tropical forest loss is so rapid, and the concentration of the world's species in these ecosystems so great (Wolf 1987), it has been suggested (e.g. Ehrlich and Ehrlich 1981, Simberloff 1986) that their current rates of loss would result in a significant proportion of all species of plants and animals becoming extinct in the next few decades, perhaps as many as 25-30% by AD2000 (Myers 1983). Other authors (Simon 1986, Lugo 1988) have contended that there is little documented evidence of species extinction, that many species survive deforestation, and that the risks inherent in tropical deforestation have been exaggerated.

The estimation of tropical forest clearance for selected countries can be seen in Table 1.1.

Table 1.1. Recent estimates of annual rates of clearance of tropical closed forests

	1981-85 rate of FAO (1988)		Sayer and Whitmore (1991)		WRI (1990)		Rate compared to FAO
	%	$10^2 km^2$	year	$10^2 km^2$	year	$10^2 km^2$	
New World							
Bolivia	0.2	8.7	-		-		-
Brazil	0.4	136.0	1987	350	1987	800	increase
Columbia	1.8	82.0	1960-84	60	-		decrease
Costa Rica	4.0	6.5	1977-83	12.4	-		increase
Ecuador	2.4	34.0	-		-		-
Mexico	1.8	47.0	-		-		-
Peru	0.4	26.0	1988	30	-		increase
Venezuela	0.4	12.5	-		1977-83	12.4	no change
Asia: continent							
Myanmar	0.3	10.1	c. 1980	60	1975-81	67.7	increase
Cambodia	0.3	2.5	-		-		-
India	0.3	13.2	-		1975-82	150	increase
Laos	1.3	10	-		-		-
Thailand	3.0	37.9	-		1978-85	39.7	increase
Vietnam	0.8	6	-		1976-81	17.3	increase
Asia: insular							
Indonesia	0.5	60	c.1985	100	1979-84	90	increase
Malaysia	1.2	25.5	c. 1985	31	-		increase
PNG	0.1	2.2	-		-		-
Philippines	1.0	9.1	1969-88	21	-		increase

Table .... contd.

	1981-85 rate of FAO (1988)		Sayer and Whitmore (1991)		WRI (1990)		Rate compared to FAO
	%	$10^2 km^2$	year	$10^2 km^2$	year	$10^2 km^2$	
Africa: west							
Ivory coast	6.5	29	-		-	-	-
Nigeria	5.0	30	-		-	-	-
Africa: central							
Cameroon	0.4	8	c. 1987	10		-	increase
Congo	0.1	2.2	-		-	-	-
Gabon	0.1	1.5	c. 1987	15	-	-	increase
Zaire	0.2	18	-		-	-	-
Africa: insular							
Madagascar	1.5	15	c. 1986	15-30	-	-	-

Sources: [FAO (1988); Sayer and Whitmore (1991); WRI (1990) quoted in Whitmore and Sayer (1992)]
Abbreviations: FAO = Food and Agriculture Organization, WRI = World Resources Institute

The New Conservation Atlas of Tropical Forests (Collins *et al.* 1991) allows the area of closed tropical forest in the Asia-Pacific region estimated to remain in 1980 by FAO (FAO, 1988) to be compared with the most recent subsequent estimate from which the maps in the atlas were prepared (Table 1.2).

Table 1.2. Recent estimates of remaining area (in $10^2 km^2$) of tropical closed forest in Asia

Country	FAO data for 1980 (a) area of forest	Recent estimate (b) area of forest	Year	change/year (%)**
India	5184	2283	1986	-9.32
Vietnam	877	567	1987	-5.04
Phillippines	951	660	1988	-3.82
Sri Lanka	166	123	1988	-3.23
Fiji	81	70	1980s	-13.58
Malaysia	2100	2005	1986	-0.75
Burma (Myanmar)	3194	3119	1987	-0.33
*Indonesia	11389	11721	1989	+0.65
Bangladesh	93	97	1986	+ 0.71
Solomon Islands	242	256	1980s	+ 5.78
Thailand	923	1069	1985	+ 3.16
Brunei	32	47	1988	+5.85
Laos	841	1246	1987	+ 6.88

Sources: [FAO (1988); Collins *et al.* (1991)] quoted in Whitmore and Sayer (1992) * The Indonesian Ministry of Forestry (1991); ** Calculated from data provided

Seven countries in Asia show a reduction and six others an increase in forest area during the 1980s. The biggest forest losses are occurring in countries like India, Vietnam and Philippines whilst the biggest increases are evident in Laos. However, these figures are flawed in a number of ways which bedevil all investigations into tropical forest clearance. One problem concerns the definition of a forest. For example, the FAO definition includes all closed tropical forests and in India this includes extensive thorn forest; the latter is excluded from the conservation atlas (Collins *et al.* 1991). This also explains the big difference between the two figures for Sri Lanka. For Vietnam, Cambodia and Laos, which all show large differences, there were also substantial problems in delimiting tropical moist forest and decisions are often subjective (Whitmore and Sayer 1992). The Philippine forests have been mapped as part of a German aid project (Forest Management Bureau 1988) and the big reduction in forest area shown for the 1980s is probably real. For the other countries in Table 1.2 there is no way of ascertaining how much of the forest decrease shown for countries like Fiji, Malaysia and Burma is real. A decrease may have also occurred during the 1980s in Indonesia, Bangladesh, New Guinea, Thailand and Brunei and the apparent increase in Table 1.2 could be an artefact attributable to differences in definition of forest. Therefore, the different estimates of tropical forest must be interpreted with extreme caution. The difficulty of obtaining good basic data may still obscure the real status of tropical forests.

1.1.2. Forestry practices

Global consumption of wood products is currently running at around 3000 million m^3 annually (Whitmore 1990). Around 55% of this is used for fuelwood; 45% is industrial forest products, two thirds of which is composed of pulp and the remaining one-third of timber. Timber exports from tropical countries are worth around US$ 7500 million per annum, ranking fifth among non-oil exports. Almost 60% of tropical forest timber products traded originate in three South-east Asian countries : Malaysia, Indonesia and Philippines (the last being of lesser importance since 1986, Whitmore and Sayer 1992). Selective logging is one of the most prevalent land uses in the humid tropics. Entrenched logging industries have existed

for decades in Southeast Asia (Abdulhadi 1975), and mechanized operations are expanding rapidly into relatively unexploited areas of the Neo-tropics (Uhl and Vieira 1989) and New Guinea (Kasenene 1987).

There are a few places where selective logging and silvicultural treatments have perhaps been undertaken successfully, and in some cases, repeat fellings have been achieved after 30-70 years without any obvious (subjectively determined) changes in the forest composition concerned, other than targeted increases in the proportion of exploitable species. Examples of apparently better management systems have been reported from Uganda (Plumptre and Earl 1984), Surinam (De Graaf 1986; Jonkers 1987), and Burma (IUCN 1987). Even in South-east Asian forest it has been indicated that changes could be made which would improve the efficiency of logging operations (Marn and Jonkers 1981; Marn 1982). It is certain that most tropical moist forests could be managed as truly renewable resources, if human intervention operated within the inherent limits of the natural cycle of growth and decay found in all forests (Whitmore 1990). However, even if management techniques are developed they are rarely put into practice. Logging is more often affected by forest concessionaires, to whom short-term profits are of the most importance, rather than by foresters, whose duty is to long-term maintenance of the resource.

Exploitation of tropical timber is increasingly a highly mechanized capital-intensive industry, particularly in South-east Asia, where logging is primarily geared towards the export market. This commercial logging in tropical forests can take a number of forms, almost all of which involve the removal of selected trees rather than the clear-felling of whole stands. The only exceptions are three operations (Colombia, Papua New Guinea, Sabah) where forests are clear-felled for wood chips. This is different from temperate forestry operations where clear-felling is much more common (Whitmore 1990). The development of tropical forestry practices rises through the scale of traditional use, commercial use of a few species, intensive use of many species, and finally (theoretically) complete utilization. In Indonesia, for example, only 10 species were used in 1900, increasing to 30 species by 1930, and over 100

species by 1945 (Hamzah 1985). The number of species used is increasing in many parts of the world. In Amazonia, up to 140 species may be logged in the eastern forests accessible both to local markets and to the populated regions of southern Brazil (Uhl and Viera 1989). On a global scale, the major sources of tropical hardwoods in West Africa and South-east Asia are a reflection of the preponderance in these areas of light- and medium-density hardwoods of high utility value. The greater proportion of high-density hardwoods over much of Amazonia will largely preclude similarly intensive harvesting for export (Whitmore and Silva 1990).

In South-east Asia up to 72 trees may be felled per ha (Whitmore 1984), although extraction levels of 20-25 trees/ha are more representative (Johns 1989). A moderate level for hill dipterocarp forest in West Malaysia commonly reaches extraction levels of 25 trees/ha (Burgess 1971). Elsewhere, for example in Gabon-Africa; White (1992) estimated that canopy loss (indicative of overall damage) averages 10% when the extraction level varies from 2-15 trunks/ha. However in regions where logging is more intensive and has been repeated several times, the forests are much changed in terms of composition and structure. In Ghana, an extraction level of only 1.6 trees/ha has resulted in 20-30% canopy loss, leaving about 3% of the ground covered by roads and loading bays (Hawthorne 1991). Removal of marketable trees is, however, only one of the effects of selective logging. Felled trees are commonly large emergents and their felling causes considerable damage to other layers of the forest (an emergent tree of > 2.5m girth will destroy around 0.02 ha on falling: Dawkins 1959). Construction of main access roads for removal of timber is generally followed by the clearing of 20-30m wide strips through the forest (this 'daylighting' is made to help drying out of the roads after rain); these main roads and their associated loading or landing areas result in the loss of 6 to 20% of the forest (Hamzah 1978, Malvas 1987). The most damaging effects arise as a result of the construction of skidroads, built by tractors to move felled trees to main access roads. These average 4m in width and may total 27km in length per km^2 of productive logged forest in Peninsular Malaysia with extraction levels about $50m^3$/ha (Whitmore 1984). Kartawinata (1978) estimates that 30-40% of a logged forest in Indonesia may be left bare of vegetation as a result of roading and dragging

activities. Incidental damage caused by construction of roads and log-loading areas, and by cut timber trees pulling down their neighbours, results in far greater losses than the removal of the target tree species. Causes of tree loss at a number of sites for selected tropical areas highlight the importance of damage to residuals (Table 1.3).

Table 1.3. Causes of tree mortality during logging expressed as % of trees lost (> 30cm girth)

	Country					
Cause of damage (killed trees)	Ponta da Castanba Brazil (1)	Nigeria (2)	S. Tekam Malaysia (3)	S. Pagai Indonesia (4) (5)	G. Tebu Malaysia (6)	Camp 92* Indonesia
Timber harvest	0.6	1	3	8	10	4.3
Construction of access roads, landing sites, etc.	60.4	25	48	46	55	49.8
Felling and dragging Remaining	39	74	49	46	35	45.9

Source: [Johns (1986a); Redwood (1960); Johns (1986b); Whitten *et al.* (1984), Burgess (1971) quoted in Whitmore and Sayer (1992)] *Anonymous (1995)

Table 1.3 shows that the incidental damage caused by mechanized logging was much higher (c 50%) than the trees killed during timber harvesting. The exception to this is in Nigeria (c 25%) where the logging systems operated are polycyclic and yields have been low (averaging 8.4 and 13.5 m^3/ha respectively in the early 1980s) (Freezailah 1984). In contrast, in dipterocarp forests of South-east Asia the stimulus to change has been the long rotation times of monocyclic systems and utilization is very high (generally more than 50m^3/ha and up to 110m^3/ha in Sabah, Malaysia) and further increases in extraction levels would certainly be detrimental to the regenerative properties of the forest (Whitmore and Sayer 1992).

Although only a small portion (1-10%, Table 1.3) of the standing trees are removed, mechanized logging generally destroys anywhere from 20 % (Crome *et al.* 1991) to

65 % of the basal areas of stands (Nicholson 1958, Johns 1988). Most damage to vegetation is incidental to the actual harvest; thus the net effect is not "selective" removal for a few timber trees, but a broader impact across a nearly random cross-section of tree, shrub and ephiphyte species (Johns 1988). Although damage can be substantially reduced by careful management practices (e.g., Abdulhadi *et al.* 1981, Crome *et al.* 1991), logged areas commonly show extensive loss of canopy cover, heavily damaged understory vegetation, altered microclimate conditions, soil compaction, increased erosion, and marked changes in the abundance of leaves, flowers and other food resources (Johns 1988, Uhl and Vieira 1989, Thiollay 1992). The consequences of these changes for forest dwelling flora and fauna are often devastating (Whitmore and Sayer 1992). However if one considers total species richness not all habitat change associated with forestry practices is detrimental. For example some vertebrate species (which may not be forest species), including some birds and primates, may be favoured by the opening up of the canopy which occurs during forest exploitation as indicated in several studies (Wilson and Wilson 1975, Kartawinata 1978, Leighton and Leighton 1983, Johns 1988) that have documented the rate at which animal and plant species recolonise forests after logging. The speed with which original animal and plant communities recolonise an area is related to the degree to which forest is disturbed and to the proximity of areas of undisturbed forests which can act as refuges for mobile species and sources of potential recolonists (Johns 1992).

The overall impact of logging operations is influenced by a range of factors; the number of trees removed; the care taken during felling operations; the extent to which timber trees are important as food sources for particular animal species; and the extent and type of post-felling silvicultural treatments applied to the logged-over forest (Whitmore 1984). A study in Sarawak, Malaysia indicated that felling trees in the direction which caused least damage, careful placement of skidroads and the restriction of tractors to them reduced damage levels by half without significantly increasing operational costs (Marn and Jonkers 1981; Marn 1982). Despite the economic importance of tropical forestry and the extent of operations in rain forest areas around the world, remarkably little effort has been directed towards examining

the environmental consequences of logging operations and how they affect the forest ecosystem. It is often assumed that tropical forestry and wildlife conservation are incompatible and that harvested forests cannot also meet conservation objectives (Sormin 1988).

1.2. Role of Amphibians in Forest Ecosystems

Primates and birds have been widely studied and examined as potential indicators of the extent of ecosystem recovery either in disturbed forest areas or logged over forests (Johns 1992). Growing interest in the integrity and diversity of forest ecosystems has prompted research into a wide range of taxa including amphibians and it has been suggested that amphibians could serve as indicator taxa for monitoring ecosystem pertubations (Vitt *et al.* 1990, Dunson *et al.* 1992, Blaustein 1994, Peachman and Wilbur 1994). Recent concern over apparent declines in amphibian populations worldwide, has also generated increasing interest in the effects of widespread management practices such as timber harvesting, on this taxon.

Amphibians are generally abundant and functionally important in most freshwater and terrestrial habitats in tropical, subtropical and temperate regions. They are the most abundant vertebrate group in many forest ecosystems, although their small size, nocturnal activities, and often fossorial habits make them relatively inconspicuous. Evidence of their abundance is found in the temperate region such as in the American forests. For example, red-backed salamander (*Plethodon cinereus*) have been recorded in the eastern deciduous forest at densities of 0.9 to 2.2 individuals/m^2 (Heatwole 1962; Jaeger 1980). The biomass of salamanders in a northern hardwood forest ecosystem in New Hampshire, USA was twice that of the bird community during the breeding season and nearly equal to that of small mammals (Burton and Likens 1975), while in southern Appalachian forest USA salamander biomass may exceed that of all other vertebrates combined (Hairston 1987). But unfortunately, there is almost no available information regarding the amphibians abundant in tropical areas. The only available information about amphibian numbers in the tropics was reported by Inger (1979) and Scott (1976). Scott (1976) has commented on the extreme scarcity of both amphibians in

Bornean forests, and of small litter vertebrates. Teams of collectors were only able to amass an average of 24.1 amphibians per day on two Bornean sites with good soil (Inger 1979) compared with the usual 50-100 specimens a day gathered by a single collector in Costa Rica and Panama.

Amphibians can occupy key trophic positions in food webs of both aquatic and terrestrial systems. As adults, they can be top carnivores, and as larvae or juveniles, they may be the major food source of many other species including birds, mammals, fish and invertebrates (Blaustein and Wake 1990). For example, around 50% of the spring diet of common black hawk (*Buteogallus anthracinus*) may comprise of riparian amphibians (Millsap and Harrison 1981). Any activity that reduces the numbers of aquatic amphibians may therefore also affect the populations of predators like common black hawk. Salamanders are also known to be prey for a diverse group of vertebrates including birds, small mammals, snakes (Brodie and Howard 1973, Brodie *et al.* 1979, Ernst and Barbour 1989) and other amphibians (Hairston 1987). Many amphibian species are important prey items for predators such as snakes and being relatively easy to capture, lacking an indigestible covering and having tissue high in protein and phosphorus (Pough 1980, Burton and Likens 1975). Amphibians also occupy a niche at the interface between aquatic wetland and terrestrial habitats and they have fast-growing aquatic larval stages specialized to exploit the high productivity of temporary pools (Wilbur 1980).

Amphibians may also play a key functional role in forest ecosystems, for example in nutrient cycling by regulating populations of soil invertebrates responsible for decomposition (Burton and Likens 1975, Hairston 1987). Terrestrial salamander species are thought to regulate prey populations of the soil microfauna (such as hymenoptera, collembola, araneida and other insect larvae (Burton and Likens 1975, Fraser 1976, Jaeger 1972) and thus, indirectly, populations of decomposing bacteria and fungi and hence productivity vital in carbon and nitrogen cycling in forest systems (Bormann and Likens 1979).

Amphibians may also serve as valuable indicators of environmental change (Wake and Blaustein 1991). Their permeable skin readily absorbs substances from the surrounding aquatic or terrestrial environment making them sensitive to pollution (Frisbie and Wyman 1991). This in addition to several aspects of amphibian natural history, including poikilothermy, exposure to both aquatic and terrestrial environments, small home ranges, high philopatry, relatively limited dispersal ability make them sensitive to environmental change and so potentially valuable for monitoring the effects of local environmental perturbations (Sinch 1990, Dunson *et al.* 1992, Blaustein 1994).

1.3. The effects of Forestry Practices on density and species richness of Amphibians

1.3.1. Introduction

Questions about the compatibility of forest harvesting practices and conservation of biodiversity are largely driven by concerns that habitat quality for many species may be degraded in intensively managed forest. The literature on relationships between common forest harvesting practices and the effects on species richness and abundance of amphibians are presented in this section. The information is mainly from the American literature that concentrates on temperate regions. Clear-cut harvesting generally has negative short-term impacts on local amphibian populations, especially salamanders (Phelps and Lancia, 1995). Most studies examining the effects of clear-cuts on amphibians have reported a higher abundance on forest control plots than in harvested plots. This pattern was consistent for both active sampling methods using area- and time-constrained searches (Bury 1983; Pough *et al.* 1987; Ash 1988; Corn and Bury 1989; Dodd 1991; Petranka *et al.* 1993, 1994; Dupuis *et al.* 1995) and most passive sampling designs using pitfalls or cover boards (Blymyer and McGinnes 1977; Enge and Marion 1986; Bury and Corn 1988, Corn and Bury 1991; DeGraaf and Yamasaki 1992 and Foley 1994).

However, the long-term effects of forest harvesting on amphibians vary and for many species these effects may be mitigated if logging practices leave adequate

microhabitat structure intact to regenerate. For example, Bury and Corn (1988) analysed pitfall capture data from 30 sites across an age sequence of young (30-76 years old), mature (105-150 years), and old growth (195-450 years) Douglas fir (*Pseudotsuga menziesii*) stands in the Cascade Mountains of Oregon and Washington. There were no significant differences in abundance among these three different age stands, except that one salamander the Ensatina (*Ensatina eschscholtzii*), was least abundant in mature stands and most abundant in young- and old-growth stands. This pattern of captures was attributed to a correlation ($r = 0.48$; $P < 0.01$) between Ensatina abundance and the low amounts of coarse-woody debris found in mature stands. In contrast, long-term effects can be significant in forest plantations, which are often associated with intensive site preparations and stand management practices that modify levels of coarse woody debris and other microhabitats. Wyman and Jancola (1992) reported a significantly higher richness (7.7 versus 2.8 species) and density (0.48/m2 versus 0.08/m2) of amphibians in naturally regenerated beech (*Fagus grandifolia*) stands than in old coniferous plantations.

The negative effects of habitat fragmentation on reptiles have been reviewed elsewhere (Raxworthy 1988), but to date there have been no detailed studies of the ecology and impact of logging on tropical amphibians. Existing studies of tropical amphibian species assemblages involve either species listings (Scott *et al.*, 1983; Cordero 1987; Zimmerman and Rodrigues 1990), single-species population and life history studies (Crump 1986; Sexton 1988; Gascon 1991) or detailed studies of community structure and reproductive ecology (Inger and Colwell 1977; Barbault 1984; Fauth *et al.* 1989). Furthermore studies of amphibians in North Borneo have focussed on species that rely on the forest streams throughout their life cycle (Inger 1969).

1.3.2. Effects of forestry practices on forest amphibian density

The effects of logging on abundance of amphibians depend upon variables such as amphibian species, elevation, forest type and age . An analysis of the results of 18 studies in the US region (Table 1.4) that examined the effects of clear-cutting on

amphibians reported total captures of amphibians on control sites had a median value 3.5 times higher than in the clear-cuts (Raphael 1988). Some amphibian groups are more sensitive to clear-cutting than others.

Table 1.4. Eighteen studies comparing amphibian captures from clear-cuts and older forest controls in the US region.

Forest region, type	Clear-cut age	Control** Captures	Method*	Principal genera	References
Western North Carolina Mixed deciduous forest (elevations: 817-1667m)	2-10 years	5x	ACS	*Plethodon Eurycea*	Petranka *et al.* 1993
Western North Carolina Mixed deciduous forest (elevations: 384-1159m)	1-4 years	4.5x	ACS	*Plethodon Eurycea*	Petranka *et al.* 1994
Northern Florida Slash pine	3-4 years	10x	PF	*Gastrophryne R scaphiopus*	Enge and Marion 1986
Northern Carolina Redwood forests	6-15 years	10x	ACS	*Bufo attenuatus*	Bury 1983
Western North Carolina	0.5-3.5 years	7.5x	ACS	*Plethodon*	Ash 1988
Oak Virginia forest	2-7 years	8.0x	PF	*Plethodon*	Blymyer 1977
New York Mixed hardwoods	7 years	2.3x	ACS	*Plethodon Notophthalmus*	Pough *et al.* 1987
Northern Carolina Douglas-fir	<20 years	3.7x	PF,ACS	*Ensatina Plethodon*	Raphael 1988
Washington and Oregon Douglas-fir	<10 years	3.2x	PF	*Ascaphus, Rana taricha*	Bury and Corn 1988
South Carolina hardwoods	2.5 years	1x	PF, CB	*Bufo, Rana Gastrophryne*	Phelps and Lancia 1995
Southern Texas hardwoods	1.5 years	1.7x	PF	*Rana, Bufo Gartrophryne*	Foley 1994
Kentucky, mixed deciduous	2 years	0.8	PF	*Bufo, Plethodon Notophthalmus*	Pais *et al.* 1988
New Hampsire, northern hardwoods	3-20 years	3.7x	CB	*Plethodon*	De Graaf and Yamasaki 1992
Vancouver Canada Douglas-fir	17-18 years	59x	ACS	*Plethodon*	Dupuis *et al.* 1995
Southern Alabama hardwoods	0.5 year	3.6x	PF	*Bufo, Rana Eurycea*	Clawson *et al.* 1996
Western Oregon Douglas-fir	<10 years	1.7x	PF	*Ensatina Plethodon*	Corn and Bury 1991
Western Oregon, headwater streams	14-40 years	3.5x	ACS	*Plethodon Dicamptodon*	Corn and Bury 1989
Maine, hemlock beech	2-11 years	2.4x	PF	*Rana, Plethodon Notophthalamus*	Pais 1987

* ACS, area constrained search; PF, pitfall traps; TCS, time-constrained search; CB, cover boards

** These values represent the relative magnitude differences in captures for control versus clear-cut stands, calculated from reported mean captures of the references. Values > 1.0 represent higher capture rates from controls.

Compared with urodeles, anurans have higher operating and tolerance temperatures (Stebbins and Cohen 1995) perhaps related to an ability to store and reabsorb large quantities of water in the bladder (up to 20-30% of their body mass: Zug 1993), and this affords them a greater tolerance of warmer and dryer microclimates in clear-cuts.

Drift-fence sampling for one year yielded information on effects of clearcutting and site-preparation intensity on pine forest herpetofauna in Bradford county, northern Florida. This study focussed on a naturally regenerated 40-year-old slash pine forest (*Pinus elliottii*) and two adjacent 3-4 year-old clearcuts subjected to different site treatments. Clearcutting and site-preparation treatments reduced amphibian (Eastern narrowmouth toad *Gasthrophyne carolinensis* and Southern leopard anuran *Rana sphenocephala*) abundance tenfold (Enge and Marion 1986). Part of the difference in amphibians'responses to clearcutting may be attributed to Bradford pine forests often containing more species dependent on water and hence being sensitive to hydroperiods, increased insolation, and decreased relative humidity. Another possible explanation for the decrease in Bradford pine forest amphibians following clearcutting is that these amphibian communities are composed of species whose main refugia are plant litter cover. Forest litter is important in providing shelter and offering a relatively cool moist condition for amphibians and has an important influence on amphibian abundance (Pough *et al.* 1987; Bonin 1991). This suggests that forest harvesting practices that minimize soil compaction and litter disruption enhance recovery rates for amphibian species associated with this microhabitat (e.g., *Plethodon cinereus*). After clear-cutting, for example, reduced leaf fall combined with increased rates of decomposition lead to a decline in forest floor organic matter. The time required for litter recovery to reach predisturbance levels can be considerable, e.g., up to 50-80 years in northern hardwood forests (Aber *et al.* 1978; Likens *et al.* 1978; Hughes and Fahey 1994). In addition, Scott (1976) mentioned that forest litter recovery was a prerequisite for repopulation of disturbed sites by amphibians.

However, the dramatic decrease in overall herpetofaunal abundance following clearcutting and site preparation of pine forests (*P. elliottii*) is not seen in more xeric pine communities in Florida. A drift-fence study in a longleaf pine-turkey oak (*Quercus laevis*) community found that amphibians were 1.7 times more abundant in a 0.5–2-year-old clearcut than in a 50–60-year-old forest (Christman *et al.* 1979). Campbell and Christman (1982) also indicated that site-preparation practices usually decrease plant litter but often increase the number of stumps and fallen logs and create more diverse habitat, therefore the species numbers also increased (Campbell and Christman 1982). In addition, clearcutting has the dramatic short-term effect of creating early successional habitat with high ground level productivity of grasses, forbs, arthropods, and small mammals (Enge and Marion 1986), potentially providing refugia and greater prey availability and therefore increasing numbers of amphibians.

Studies examining the relationship between amphibian abundance and forest age report varied results including no species-seral stage associations (Bury and Corn 1988; Aubury *et al.* 1988, Gilbert and Allwine 1991; Diller and Wallace 1994), a few species with weak relationships to forest age (Welsh 1990, Welsh and Lind 1991; Raphael 1988; Grant *et al.* 1994), and convincing, repeatable increases in abundance with forest age for nearly all species examined (Petranka *et al.* 1993, Dupuis *et al.* 1995). Part of the explanation for why forest age is not a stronger and more consistent predictor of amphibian abundance may be that it is not age per se that determines species distributions but rather the associated microhabitats that vary in relation to forest age during succession (Aubry *et al.* 1988; Welsh 1990). Thus, forest seral stage is only an indirect measure of the actual microhabitat elements (e.g., coarse woody debris, foliage height diversity, canopy cover, litter type and depth, and cool, moist equitable conditions) that determine whether or not a site is a suitable habitat for a given species.

For example, several studies exploring forest age effects have found a statistically significant relationship between salamander captures and the presence of coarse woody debris (Bury and Corn 1988; Aubry *et al.* 1988, Raphael 1988; and Petranka

et al. 1993). Petranka *et al.* (1993) found that clear-cuts (<5 years) and old stands (>120 years) had the highest amounts of coarse woody debris (a preferred amphibian microhabitat) which may relate the abundance of amphibians to forest age.

Bury (1983) reported that the abundance and biomass of the Californian amphibian *Bufo attenuatus* was related to the age of redwood forests in northwestern California. This species was 10 times more abundant in pristine, old growth forest (mean 34 amphibians per 0.125-ha plot, range 18-61) than in similar sites logged 6 - 14 years previously (mean 3.3 amphibians per 0.125-ha plot, range from 1-9). This evidence suggests that logging has a long-term effect on resident herpetofauna of redwood forests. Apparently, opening of the canopy favors a few species to the detriment of the majority of forest-dependent species. Bury attributed the differences to microclimatic changes, such as greater light penetration on opened forest floor, leading to increased soil temperatures and evaporative water loss, and greater variation in temperature and humidity. Such conditions could be unsuitable for *B. attenuatus*, a species of which Hendrickson (1954) proposed should serve as a more sensitive indicator of vital limits and of optimum climate than would most other anurans.

Corn and Bury (1991) reported that the number of salamander species (*Ensatina* and *Plethodon*) caught by pitfall traps in the control plots was almost double that in the clear-cut plots of 10-year old Douglas-fir in Western Oregon. Similarly, Aubry and Hall (1991) found that roughskin newts (*Taricha granulosa*) were four to five times more abundant in old-growth forests than in young-logged stands. Although some other studies here found the reverse eg. Gilbert and Allwine (1991). Old-growth forests probably offer important protected refugia and suitable temperature-moisture regimes for terrestrial newts in the form of fallen logs, bark, leaf litter, and decaying wood.

A study in northwest California Douglas-fir (*Pseudotsuga menziesii*) forests reported that three (Del Norte salamander, *Plethodon elongatus*; black salamander,

Aneides flavipunctatus; and Ensatina, *Ensatina eschscholtzii*) of the six species of salamanders were associated with older forests, while both of the anuran species (western toad, *Bufo boreas*; and Pacific tree anuran, *Hyla regilla*) were more abundant in younger stands (Raphael 1988). Another study in northwest California indicated that five salamander species (mainly plethodontids from the genera *Plethodon* and *Desmognathus*) had significantly higher abundance on older sites and three revealed a low but significant linear relationship with forest age (tailed anuran *Ascaphus truei*, $r^2 = 0.29$, $P < 0.05$; Del Norte salamander *Plethodon elongatus*, $r^2 = 0.24$; Torrent salamander *Rhyacotriton variegatus*, was completely absent from young stands, $r^2 = 0.30$; Welsh 1990). These and previous results (Raphael 1988 and Welsh 1990) for Del Norte salamander contrast with those of Diller and Wallace (1994) who found no relationship between amphibian abundance and forest age. However, it is unclear whether they sampled the same age classes.

Petranka *et al.* (1993) reported that captures of amphibians (*Plethodon* and *Eurycea*) in mature forest stands were five times higher than those in recent clearcuts in the Southern United States and estimated that clearcutting in U.S. National Forests in western North Carolina could result in the loss of c. 14 million amphibians. However, such a great loss is based on the assumption that most salamanders die after complete timber removal. Ash and Bruce (1994) reject this assumption and cite Madison and Shoop's (1970) data on Jordan's salamander (*Plethodon jordanii*) moving 60m in a 12-h period to argue that plethodontids are capable of long distance dispersal, despite their relatively small home ranges. Indeed, mark-recapture data for the Peaks of Otter salamander (*Plethodon sp.*), an extremely sedentary animal in undisturbed habitats, also suggests salamanders are capable of moving off-site after clear-cutting in the Jefferson National Forest of Virginia (Kramer *et al.* 1993). However, Jaeger (1980) suggests that salamanders will migrate vertically (below cover objects and eventually underground), and not horizontally when physiologically stressed by dehydration as might occur after canopy removal. Once underground, feeding and reproduction opportunities are limited (Jaeger 1972; Fraser 1976), the environment may become increasingly toxic

to osmoregulatory mechanisms (Frisbie and Wyman 1992), and the animals, Petranka argues, slowly perish.

Whether lowered abundance of amphibians in recent clear-cuts is caused by death, dispersal or hibernation is unclear. For example, some salamander species can remain inactive in underground retreats, at depths down to 90cm (Grizzell 1949), for long periods of time as "sit and wait exploiters" of equitable surface conditions for feeding (Feder 1983). In this case the chances of being captured are reduced because most amphibian sampling protocols require at least partial surface activity by the animals (e.g., pitfall traps, cover object serving, night censusing, quadrat litter plots). The question of resident animals' fate after clear-cutting has potentially important ramifications. If a reservoir of subterranean salamanders persists on site, they will accelerate the rate at which the population in the regenerating stand recovers to predisturbance levels. In the absence of such a reservoir, population growth will depend exclusively upon horizontal immigration rates from adjacent habitats and is likely to be much slower.

Quadrat sampling conducted at Nanga Tengalit (North Borneo rain forest) and Bukit Lanjan (30-year-logged forest of North Borneo) yielded information essentially only on terrestrial *anurans* (mainly from genus *Bufo* and *Rana*). The mean number of *anurans* per quadrat was larger at the Nanga Tengalit rain forest site (0.76/quadrat, n = 402 quadrats) than in Bukit Lanjan logged forest (0.18/quadrat searched, n = 479 quadrats) (Inger 1979). The differences could be attributed to the differences in temperature and humidity between sites and the susceptibility of *anurans* to desiccation at the higher temperatures.

1.3.3. Effects on species richness

Results are less conclusive for the effects of logging on species richness than for abundance, with fewer studies reporting statistically meaningful comparison of clear-cuts to older stands for multiple species. Petranka *et al.* (1993, 1994) found significantly fewer amphibian species on clear-cut plots in the southern

Appalachians, USA (approximately one half and one third of the richness of mature stands at high and moderate elevations, respectively). Similarly, for a community of stream amphibians in Western Oregon forest, Corn and Bury (1989) reported that only 1 of 20 (5%) streams in logged stands contained all four of the species studied compared with 11 of 23 (47.8%) streams in uncut forests. However, Raphael (1991) found no significant differences in patterns of species richness between clear-cut and older seral stages, while several others found differences in species composition after cutting and a trend toward greater species richness on control plots (Bury 1983, Enge and Marion 1986; Corn and Bury 1991); but these differences were not statistically significant.

Differences in taxa represented in the various data sets may contribute to discrepancies across studies. For example, Petranka *et al.*'s (1993, 1994) results are based primarily on 10 species of Plethodontidae, a family whose members dominate the southern Appalachian salamander fauna. In contrast, Enge and Marion's (1986) data set consisted of 18 amphibian species, only two of which are lungless salamanders and none of which are entirely terrestrial (life history traits common to plethodontids that may predispose species to sensitivity to changes in microhabitat and microclimate after timber harvesting).

In the Pacific Northwest, amphibian populations of several species have apparently become locally extinct, and the ranges of others have become drastically reduced (Blaustein and Wake 1990). Fifty-four percent of the native Oregon amphibian species are listed as Sensitive to logging, and 46 percent of native amphibians of Washington State in the Special Concern category (Walls *et al.* 1992). The increasing loss of old-growth forest habitat is likely to have detrimental effects on the herpetofauna of the Pacific Northwest. Similar effects are apparent in eastern North America where, for example in Central New York State, it was reported that amphibian species richness in recently disturbed habitats was much lower than in old-growth habitat (Pough *et al.* 1987).

Changes in numbers of salamander species during succession have been studied in the hotter and drier lower elevation forests in western North Carolina, North America. The results indicated that mean number of species increased significantly with the stand age class (<40, 40-79, 80-119, and >120 years), and effects appeared more pronounced with age on dry stands (lacking streams) and reached peak levels in forests of 120-200 years old. No salamanders were captured on recent clear-cuts in dry stands (Petranka *et al.* 1994). The authors concluded that salamander populations in the drier forests of western North Carolina recovered more slowly than populations at higher elevations where the climate is cooler and moister.

Grant *et al.* (1994) examined the effect of forest age on amphibian species richness in pine plantations in the coastal plain of South Carolina by using drift-fence arrays. The study results showed that overall species richness (seven salamander species and 14 anurans) was highest in the oldest plantations and lowest in the first year plantations. The authors report that intermediate-aged stages of loblolly pine (*Pinus taeda*) regeneration yielded the highest diversity (Shannon and Simpson's index) of amphibians. Their conclusions were driven largely by the disproportionate abundance of two toad species (*Scaphiopus holbrooki* and *Gastrophyne carolinensis*) in the youngest and oldest stands, which caused species evenness to be greatest in intermediate-aged stands.

Finally, with regard to amphibian species richness, an in-depth examination, 160 sites among four age classes, from clear-cutting to old growth (>250 years) in northwestern California by Raphael (1991) found no trend in species richness associations with forest age.

1.3.4. Effects of logging on stream dwelling communities

Effects on abundance of amphibians

The effects of logging on stream habitats and on invertebrate and vertebrate faunas are complex (Murphy and Hall 1981). Long-term effects of logging on streams seem

to be generally negative (Scivener and Andersen 1984). Vertebrate populations are usually reduced in abundance in streams in managed second-growth (Murphy *et al.* 1981) where food quality may be poor and habitats unstable (Sedell and Swanson 1984). Low-gradient streams may retain fine sediments (Murphy *et al.* 1981; Hawkins *et al.* 1983), and streams in managed forests will suffer from reduced input of large organic debris (Swanson and Lienkaemper 1978; Bryant 1985).

A few studies in the Oregon Cascades and Coast Range of North USA on the impacts of logging on streams include data on amphibians (Murphy and Hall 1981; Murphy *et al.* 1981; Hawkins *et al.* 1983; Conner *et al.* 1988). In all cases the biomass of Pacific giant salamanders (*Dicamptodon ensatus*) was highest in streams in recent clearcuts and lowest in streams in old-growth forests. However, the abundance of amphibians for stream side areas was greater in the primary logged stands than in the 3-year old logged stands in New England (DeGraaf and Rudis 1990) and a similar result was also reported in the Pacific Northwest (McComb *et al.* 1993; Dupuis *et al.* 1995).

Bury and Corn (1988) found a mean of 3.7 amphibians/m^2 and a biomass of approximately 10g/m^2 in forested headwaters of the Oregon Coast Range, USA while in the nearby 5-year old stands the mean amphibians were reported as less than 1 amphibian/m^2 with the biomass around 2.5g/m^2. Effects of logging on stream amphibians in Oregon Coast Range were further examined by Corn and Bury (1989). They compared logged versus nonlogged reaches of streams in areas where stream gradients were high and low. In logged situations, they found high densities of amphibians only in high-gradient portions of the streams. In uncut stream reaches, amphibians were found both in high- and low-gradient areas. They attributed these results to the increased levels of fine sediment present in low-gradient, logged areas. Fine substrates appear to fill cracks and crevices in the streams, thereby altering critical microhabitats used by these animals.

Corn and Bury (1989) have conducted the only comparative study that comprehensively sampled the amphibian community in streams passing through

logged and unlogged stands. In the Oregon Coast Range they compared the amphibian fauna from 23 streams in uncut, naturally regenerated forests aged 60 to over 400 years old to 20 biophysically matched streams in logged stands aged 14 to 40 years. The logged streams had signifiantly greater overhead vegetative cover than unlogged streams, despite the lack of buffer strips, because of dense pioneer hardwood understorey regeneration. The density and biomass of amphibians was significantly higher in unlogged streams , with the ratio of mean density in unlogged stands to mean density in logged stands ranging from 2.0 to 7.2. The most important physical difference reported between the treatments was a greater amount of fine sediments in logged streams, an input that can degrade microhabitat quality of larval amphibians by reducing interstitial space within the stream substrate.

Improper construction of logging roads and certain logging activities can lead to sedimentation (Packer 1967; Bormann *et al.* 1974; Patric 1976; Campbell and Doeg 1989; Swanson *et al.* 1990), especially in low gradient streams that do not have enough current to flush out accumulated material (Beschta 1978; Murphy and Hall 1981; Noel *et al.* 1986; Corn and Bury 1989). Therefore, if sedimentation is a major factor impacting the ecology of stream amphibians, then the effects are likely to be relatively long lived in low gradient streams. Murphy *et al.* (1981) found increased sedimentation on low gradient streams in clear-cuts and second growth forest but noted no accumulation of finer sediments in high gradient streams. Furthermore, a negative correlation between the amount of fine sediment and the density of Pacific giant salamanders was reported by both Hawkins *et al.* (1983) and Conner *et al.* (1988). In other words, increased fine sediment leads to a reduced density of Pacific salamanders.

Effects on species richness

Raphael (1991) found no significant differences in patterns of richness between clear-cut and older seral stages, while several others found differences in species composition after cutting and a trend toward greater species richness on control plots (Bury 1983; Enge and Marion 1986; Corn and Bury 1991); but these were not statistically significant.

McComb *et al.* (1993) and Dupuis *et al.* (1995) reported that there was no significant difference in species richness between streams flowing through Unlogged stands and Logged stands in the Pacific Northwest. Studies from Oregon that investigated the influence of logging practices on the Pacific giant salamander (*Dicamptodon sp.*) (Murphy and Hall 1981; Hawkins *et al.* 1983) reported a greater biomass of this amphibian species in stream sections passing through recent clear-cuts versus old- growth controls, but the differences in species richness were not statistically significant.

Bury (1991) reported no significant differences in species richness of amphibians for 59 streams sampled from young, mature and old growth stands in naturally regenerated forests throughout southern Washington and Western Oregon. Similarly, Corn and Bury (1989) found no significant differences in species richness for three stream species (tailed anuran *Ascaphus truei*, Dunn's salamander *Plethodon dunni* and Pacific giant salamander *Decamptodon sp.*) when 43 stands from the same region were grouped into young (<25 years, all logged) and old (>25years) forest stands.

Effects on particularly sensitive species

Welsh and Lind (1991) found that salamanders favoured cool moist forest conditions associated with ground vegetation cover and decayed woody debris. Furthermore, they reported that the amphibians were associated primarily with downed woody material such as logs, snags, and bark. Such cover or other types of physical habitat complexity can provide refuge from unsuitable temperature and moisture conditions (for example, Bury, 1983; Maiorana, 1977). In Oregon, these amphibians are considered Sensitive species to loss of large, decaying, downed snags from timber management practices (Marshall *et al.* 1992).

Other studies suggest negative logging impacts on aquatic amphibians, but these are largely anecdotal (Metter, 1964; Bury, 1983; Nussbaum *et al.*, 1983). Most studies examining the effects of logging on streams consider only the immediate effects of

clearcutting, such as effects of the open canopy. Canopy cover (% shading) over streams traversing clearcuts is transitory and regenerates to pre-logging levels in approximately 10 years in the Oregon Coast Range (Beschta *et al.* 1987; Andrus and Froehlich, 1988). Forests with 60-year rotation ages would have only about 17% of the surface area in the open canopy stage at any time. Because intensively managed forests will contain little old-growth, survival of amphibian populations in areas of timber harvest may depend on species' ability to reinvade and reproduce in streams in second-growth forests.

Tailed *frogs* (*Ascaphus truei)* are one of the most primitive of living *frogs* and are endemic to the Pacific Northwest (Nussbaum *et al.* 1983). They are the species most likely to be affected by old-growth habitat loss and degradation since they are intimately associated with the cool fast-flowing streams common in old growth forested areas (Raphael, 1988). A number of studies have shown *A. truei* to be restricted to mature old growth forest and absent in logged sites (Bury 1983, Aubry and Hall 1991). Bury and Corn (1988) considered *A. truei* to be "sensitive to timber harvest" which destroys the cool flowing streams that they require for breeding purposes and larval development. Thomas *et al.* (1993) reiterate the importance of maintaining cool stream temperatures and reducing sedimentation as protection measures for this species. They recommend headwater stream protection through buffers as mitigation for this anuran. Tailed *frogs* were rated to have a medium to high risk viability in four of the five forest management options considered by Thomas *et al.* (1993).

Densities of Pacific giant salamanders (*Dicamptodon ensatus*) and Olympic salamanders (*Rhyacotriton olympicus*) were positively correlated with stream gradient in logged stands, but not in unlogged forests, suggesting that the disruptive effects of increased sedimentation are greatest in low-gradient streams.

1.4. Thesis Aims and Objectives

Relatively few studies have focussed on the impact of logging on amphibians. Those that have been undertaken have related mainly to temperate regions where the logging system is usually clear felling and have tended to concentrate either on single species or on single genera. Studies on the impact of selective logging on anurans have not been carried out in the tropical region of Indonesia. Therefore, this study will attempt to address this gap in our knowledge about the impacts of logging on elements of biodiversity by comparing the amphibian community in unlogged and logged areas of Bornean rainforest.

Specifically this study will (a) compare species richness of anurans between unlogged and two types of logged forests; (b) compare species abundance and similarity between unlogged and two types of logged forests; (c) relate any observed differences to habitat and microclimate differences in unlogged and two types of logged forests.

The practical objective of this study is to provide detailed information on the impact of logging on the tropical forest environment and to use this to assist in the development of policies that will integrate sustainable use with biodiversity conservation.

1.5. Thesis Structure

The thesis comprises nine Chapters:

Chapter One gives a general introduction to the thesis, including background of the study covering the status of tropical forestry and forestry practices, role of amphibians and a literature review on forestry practices and amphibians. This chapter also outlines the aims and objectives of the study.

Chapter Two provides a review of amphibian biology, taxonomy and global status with special reference to the ecology of Bornean anurans.

Chapter Three describes the study site and logging at this site in Central Borneo, Indonesia with a general outline of logging management and forestry regulation and a short introduction to the UK-Indonesia Tropical Forest Management Project (TFM Project) in cooperation with Kayu Mas Forestry Logging Concession. A description of the project site is also given in this chapter. This chapter also outlines the logging system and its impact.

Chapter Four details the research methodology and describes the sampling regime and instrumentation required for this study used to survey anurans.

Chapter Five describes the impact of logging damage on vegetation and the difference in vegetation cover between unlogged and logged sites. This chapter also outlines the microclimate data in the study sites.

Chapter Six contains the results and discussion of studies on species richness and composition, species presence and absence, abundance of terrestrial anurans, and evaluates patterns of richness and abundance in relation to environmental variables according to the pitfall trapping method.

Chapter Seven discusses the same topics and analysises the data in a similar way as presented in Chapter six according to the searched quadrat method.

Chapter Eight presents the results regarding species richness and composition, species presence and absence, distribution and niche, abundance of riparian anurans and analysis patterns of richness and abundance in association with environmental variables according to the riparian research method.

Chapter Nine deals with conclusions and outlines several aspects of the amphibian-forestry relationship in need of future research. Management recommendations relevant to conserving amphibians in relation to forestry practices are also offered.

CHAPTER II

REVIEW OF AMPHIBIAN BIOLOGY AND ECOLOGY WITH SPECIAL REFERENCE TO BORNEAN ANURANS

2.1. Introduction

For well over 250 million years amphibians have been exploiting habitats in moist tropical environments, where the bulk of them remain. Either most living amphibians are restricted to such environments or they belong to groups of species that have representatives with ranges extending into the tropics (Duellman and Trueb 1994). Biological and ecological knowledge of which species occur in which areas are fundamental to an understanding of the intricacies of biological diversity. In order to place the knowledge of amphibian biology and ecology in perspective, a brief classification of amphibians is presented with special reference to Bornean anurans. In addition, the ecology and global status of these anurans are also reviewed.

2.2. Classification of Amphibians and Bornean Anurans

2.2.1. Amphibians of the world

The Amphibia are cold-blooded vertebrates typically possessing limbs instead of fins, having soft, moist skin without hair, feathers or external scales. There are three orders of living Amphibia: the *Urodela* (*Caudata*), *Anura* and *Apoda* (*Gymnophiona*) (Berry 1975). The *Urodela* include 3 suborders : Sirenidae (other eel-like salamanders from the USA), Cryptobranchoidea (aquatic north American and Asian urodeles) and Salamandroidea (major group of small terrestrial salamanders like *Ambystoma*). The *Urodela* include the newts and salamanders which have retained the generalized characters such as limbs (rarely lost) and tail, and a moderately elongate body. Teeth are also retained in both jaws (sometimes absent). The *Anura* which comprise only the frogs and toads have, however, become greatly modified and are not generalized. The legs are retained in all cases, but they

are greatly enlarged in proportion to the arms for hopping and leaping. The body is also greatly modified and has become especially shortened, the number of vertebrae being greatly reduced. The teeth are, with rare exceptions, lost in the lower jaw and the tail has been lost in the adult, but retained in the tadpole stage.

Anurans have colonized many varieties of habitats. They have a varied life history pattern, and while normally the anurans require a free swimming tadpole stage, many forms pass the larval stage entirely within the egg, the young anuran emerging with arms and legs developed and the tail reduced or lost. Generally the eggs are deposited in water, in the form of strings (toads), in clusters (frogs), or in foam-nests (tree-frogs of the family *Rhacophoridae*). Some anurans lay their eggs in shrubs and tree-holes, or on the ground in holes, depressions, or excavations (Inger 1966).

The *Apoda* have become highly specialized for a subterranean life although large larval stages may be found beneath stones and submerged debris in streams. Adaptations for the burrowing habit include the loss of limbs and tail, the much elongated and snake-like body, the serpentine-like progression, and the reduced eyes. Like the anurans, the mode of life history varies in the *Apoda*. Eggs are usually fertilized and may be deposited in moist places under rocks, in moist burrows near stream banks or they may be retained in the oviduct, the young being born in an advanced larval stage. The *Apoda*, commonly known as caecilians, are less familiar to most people because of their subterranean habitat. The fact that only a single family is recognized at the present time, suggests that they all originated from and have diversified from a monophyletic group (Berry 1975). Two species of genus *Ichtyophis* are known from Kalimantan, the unicolored *monochrous* (Bleeker) and the striped *glutinosus* (Linnaeus) (Inger 1966).

2.2.2. Amphibians of Southeast Asia

In Peninsular Malaysia, about 80 or more species, representing 6 families and 20 genera of anurans and caecilians have been recorded (there are no urodeles in Malaysia). Despite the abundance in number and species very little has been published or is known about the Indo-Malay amphibian fauna. These include a few

articles in the Malayan Nature Journal (Berry 1975), the work of Boulenger (1912), and Smith (1930). Inger's (1966) work on the Amphibia of Kalimantan remains authoritative for the Indo-Malay region.

At least 89 species of anurans occur in Kalimantan, Indonesia. They fall in five families that are widely distributed in Southeast Asia: *Pelobatidae*, *Bufonidae*, *Microhylidae*, *Ranidae*, and *Rhacophoridae* (Inger and Stuebing 1989). The most widely held opinion on the evolution of anurans places the *Ranidae* and *Rhacophoridae* (sometimes called Oriental tree frogs) close together, although their separation goes back at least 75 million years (Matsui 1975). The *Microhylid* (narrow-mouthed anurans) are considered to be an earlier offshoot of the same line that gave rise to the *Ranidae* and *Rhacophoridae*. The other two families represent quite distinct and separate lines of descent (Inger 1966).

Each of these families is subdivided into genera. In Kalimantan there are 4 genera of *Pelobatidae*, 4 genera of *Bufonidae*, 5 of *Microhylidae*, 5 of *Ranidae*, and 4 of *Rhacophoridae*. As is evident from the checklist at the end of this section (see table 2.1), four genera account for about half of the total species list: *Rana*, *Amolops*, *Rhacophorus*, and *Ansonia*.

New species of anurans are still being discovered. Late in 1986 Inger (pers.comm.) found one new species (*Rana ingeri*) in the forest of Danum Conservation Area near Lahad Datu, and in 1987, he found another (*Bufo juxtasper Ingeri*) in Mendolong forest area, North Kalimantan. Recently, two new species of ranids were found in the study site at Kalang River, Central Kalimantan, Indonesia in January 1995 (Taufik, pers.obs). The two species of ranids have been identified and given names by RF Inger, the Herpetologist from the Field Museum of Natural History, Chicago, USA, as *Rana asperata* and *R. rhacoda*. The description of these two new species was published in the Raffles Bulletin of Zoology 1996 (Inger and Taufik 1996). The known anuran fauna of Kalimantan continues to expand at a steady rate and the number may be expected to increase, especially as faunal exploration of Kalimantan proceeds. Finally, as study of certain species proceeds, what is at present called a

single species may turn out to be two or more closely related species. That has been the recent history, for example, of species of *Amolops*, the torrent frogs. Another likely candidate for this kind of splitting is the Large-Eyed Litter Frog, *Leptobrachium montanum*. The Kinabalu population of this frog has a white eye ring not found in other populations and has a distinct belly pattern (Inger and Stuebing 1989).

In some of the larger genera, certain species seem to form related groups, particularly when the anatomy of adults and tadpoles and their behavior are examined. For example, in the genus *Rana* the species *blythi*, *kuhli* and *ingeri* constitute one such group of similar species. Among their unusual features leading to this conclusion as stated by Inger (1966) are: (1) males are larger than females, which is the reverse of size relations in most anurans; (2) males do not have vocal sacs and do not call; (3) males have enlarged fang-like bony projections at the front of the lower jaw. Their tadpoles are very similar down to fine details of structure.

The two guardian frogs, *Rana palavensis* and *R. finchi*, constitute another species group characterized by reduced webbing and, most remarkable, the habit of males in guarding the eggs and then transporting the tadpoles to water. In fact, these two are so similar that they were once thought to be races of a single species (Berry 1975).

Tree frogs of the genus *Rhacophorus* also form several groups of related species. The medium-sized to large species having fully webbed fingers, Wallace's Flying Frog (*Rhacophorus nigropalmatus*), the Harlequin Tree Frog (*R. pardalis*), and the Jade Tree Frog (*R. dulitensis*), all have smooth-edged flaps of skin along the outer edge of the fore arm, all have the ability to parachute, and all have tadpoles that live in small pools of standing water (Inger and Stuebing 1989). Two small tree frogs, *Rhacophorus gauni* and *R. bimaculatus*, form a second group. These small species have only partially webbed fingers and have pointed projections of skin near the heel. Their tadpoles live in small, clear, rocky-bottom streams.

2.3. Ecology of Bornean Anurans

2.3.1. Ecological classification

There are two major groups of anurans in Kalimantan classified on the basis of their general habits and habitats (Inger and Stuebing, 1989). One group, the smaller, consists of species closely associated with man (paddy fields) or other disturbed habitats such as town gardens, cultivation and roadside ditches. Eight species characteristic of disturbed habitats can be grouped in this category: Green Frog (*Rana erythraea*), Taiwanese Frog (*R. granulosa*), Mangrove Frog (*R. cancrivora*), Grass Frog (*R. limnocharis*), Cricket Frog (*R. nicobariensis*), Four-Lined Tree Frog (*Polypedates leucomystax*), Banded Bullfrog (*Kaloula pulchra*), and Plain Bullfrog (*K. baleata*). Not only do these species live close to people, but they also seem dependent on people to create those environmental conditions in which they thrive: open spaces, disturbed ground, or puddles of standing water.

None of these species are considered typical of primary rain forest, although *Rana nicobariensis* and *Polypedates leucomystax* are abundant in secondary growth and penetrate the edges of primary forest. All of these frogs breed in standing water; ditches, small ornamental pools in gardens, rain-filled depressions in villages and towns, and of course, flooded paddy fields (Bharata, 1989). Only one of them, *Polypedates leucomystax*, regularly perches on tall grasses and other vegetation. Two are burrowers-*Kaloula pulchra* and *K. baleata*. *Rana granulosa*, *R. erythraea*, and *R. cancrivora* rarely move more than a few metres from water, whereas *Rana limnocharis* and *R. nicobariensis* often move through damp grass and shrubbery some distance from water.

The second group comprises the majority of Bornean anuran species which are confined to forests or their edges. For example, the Saffron-Bellied Anuran (*Chaperina fusca*) and the Slimy Toad (*Bufo biporcatus*), may occasionally survive in shaded gardens. But otherwise they are confined to forest. The details of their life styles vary widely, but four life history patterns can be identified. These are described below:

a. Species that are riparian throughout their life cycles. These anurans never leave the banks of streams. Their tadpoles develop in creeks and rivers. The newly transformed froglets climb out of the water on to the banks and they feed and grow into adults that remain close to water where they feed and breed. Anurans of this type include the Giant River Toad (*Bufo juxtasper*), the Black-Spotted Rock Frog (*Staurois natator*) and the Blyth's Frog (*Rana blythi*). Anurans such as the Greater Swamp Frog (*Rana ingeri*) that are confined to marshy areas can also be grouped into the riparian species (Boeadi 1990).

b. Species of anurans that are found along stream banks and live away from water courses only as juveniles. The Spotted Stream Frog (*Rana signata*) and Poisonous Rock Frog (*Rana hosei*) which is a common species in Kalimantan, is a good example of this mode of life. According to Inger and Stuebing (1989), the Signata tadpoles develop in quiet areas of small streams. When they transform, the tiny froglets (about 1 cm long) may move a considerable distance into the forest where they feed. As they reach adult size, they return to a stream as their original habitat.

c. Species that use streams only for breeding. Juveniles of the Brown Tree Toad (*Pedostibes hosei*) emerge from side pools of streams and spend the rest of their life in the forest. They return to streams just to breed. Some common species of Bornean anurans that have this type of life are the Smooth Guardian Frog (*Rana palavanensis*), the Large-Eyed Litter Frog (*Leptobrachium abotti*), the Bornean Horned Frog (*Megophrys nasuta*), and the White-Lipped Frog (*Rana chalconota*) (Iskandar 1989).

d. Species that wander widely through the forest at all ages (not tadpoles) and are seen along stream banks only as accidental visitors. Most of the anurans of this group lay eggs in small pools of water on the forest floor, but some use water-filled tree holes and a few even lay their eggs under floor litter where there is no standing water but humidity is high (Inger 1966). Terrestrial forest species, such as the Saffron Bellied Frog (*Chaperina fusca*) and the Yellow-Bellied Puddle Toad

(*Occidozyga laevis*) also breed in such pools (Matsui 1986). Several species of tree frogs that belong to this group and live up in the canopy, including Collett's Tree Frog (*Polypedates sp.*) and Red-Legged Tree Frog (*Rhacophorus rufipes*), descend to deposit their foam nests at the edge of such pools or attach them to low overhanging vegetation.

2.3.2. Mode of life and breeding strategies

The mode of life cycle has little correlation with the vertical zone favoured by an anuran (Iskandar, 1989). There are arboreal and terrestrial species in each of these life cycle types. The anurans that live at ground level use a variety of substrates. For example, the Greater Swamp Frog (*Rana ingeri*) usually sits on mud. Whereas its larger relative, the River Dwelling Blyth's Frog (*Rana blythi*), most often sits on sand or gravel.

At least some species of the mainly shrub-dwelling anurans of the genus *Philautus* lay 6-12 relatively large eggs under dead leaves, in cavities under logs, or under moss (Kurniawan 1987). The embryo develops, not into a free-swimming tadpole, but into a tiny froglet with a tail. The froglet remains within the egg capsule until all the internal organs have reached the condition at which a "normal" tadpole would transform into a young anuran. Anurans with this type of life cycle are, of course, completely independent of water bodies. They do particularly well in hilly forests, such as those of Mt. Kinabalu, where a blanket of spongy, wet moss covers everything.

Arboreal anurans also vary in the perches they use. Several species, such as the Red-Legged Tree Frog (*Rhacophorus rufipes*) and File-Eared Tree Frog (*Polypedates otylophus*), perch on leaves of shrubs and young trees, whereas others, particularly larger species such as the Brown Tree Toad (*Pedostibes hosei*) perch on twigs and branches or even trunks of small trees (Boeadi 1990). The habits of tree anurans are very difficult to study, because they usually feed and rest at heights of > 5-6 m. When standing on the ground tree anurans can only be seen up to a certain height (less than 5 m), the branches and leaves hinder observation and make viewing

difficult. But anurans do live higher up because they can be heard from the upper canopy.

Arboreal anurans that use water-containing holes appear to recognize the difference between large and small trees (Inger and Stuebing 1989). For example, the small Tree-Hole Toad (*Metaphrynella sundana*) uses holes in trees or lianas having trunks no larger than 10-12cm. In contrast, the larger Red-Legged Tree Frog (*Rhacophorus rufipes*) was observed to lay its eggs in water - bearing cavities in the holes of trees having trunks greater than 30cm in the Permanent Sample Plot on December 1994 (pers.obs). The Brown Tree Toad (*Pedostibes hosei*) which is commonly found in Kalimantan, not only uses holes in large tree trunks but also sometimes lays its eggs in the "tanks" formed by the fused buttresses of Meranti *Shorea sp* (Tirta 1990).

2.3.3. Habitats of tadpoles

The microhabitats of tadpoles, though not as diverse as those of adults, cover a wide range. Despite the necessity for all of them to live in water, there is a lot of specialization among kinds of tadpoles in terms of particular types of aquatic environments (Inger and Stubing, 1989). Even among those that live in streams, there are major differences. For example, some live only in swift portions of rocky streams. Tadpoles of the Yellow-Legged Torrent Frog (*Amolops jerboa*) cling to rocks in the strongest currents by means of an abdominal sucker. The same microhabitat is used by tadpoles of the Slender Toad (*Ansonia sp.*), but in this species the clinging device consists of expanded lips (Smith, 1930).

Accumulations of dead leaves form in forest streams wherever the topography of the stream bed creates eddies, and these leaf drifts constitute the main microhabitat of tadpoles of Blyth's Anuran (*Rana blythi*), the Brown Tree Toad (*Pedostibes hosei*) and the Spotted Stream Anuran (*Rana signata*). These kinds of tadpoles are sometimes found in quiet side pools, that is, shallow, sheltered areas usually cut off from the main current of a creek by gravel bars or rocks (Nawangsari, 1988). These quiet pools, with their slightly silty bottoms, are the main habitat of another group of

tadpoles, particularly those of the White-Lipped Anuran (*Rana chalconota*), and the Crested Toad (*Bufo divergens*). There are also some stream tadpoles, for example, those of the Large-Eyed Litter Anuran (*Leptobrachium sp.*), that seem to do well in a variety of microhabitats no matter what the current strength or bottom type (Inger and Stuebing 1989).

Silty pools on the forest floor are the main type of environment for tadpoles of several tree anurans and those of several inconspicuous, dwarf, terrestrial species, such as the Bornean Narrow Mouth Anuran (*Mycrohyla borneensis*). Tiny pools of water, sometimes no more than 1cm deep and 10cm across, in rotting logs or even on large dead leaves provide suitable habitats for tadpoles of the Red-Sided Sticky Anuran (*Kalophrynus pleurostigma*) and of one of the small forest floor toads of the genus *Pelophryne.* Usually the water in a pig wallow will hold two or even three kinds of tadpoles, a tree anuran tadpole and one or two types of tadpoles differ greatly in size and in mode of feeding. Tree anuran tadpoles are at least twice as long as the others and feed by grubbing around in the decaying organic matter on the bottom whereas the small tadpoles of the dwarf species hover in mid-water and filter out food particles as they pump water through their mouths and over their gills (Inger, 1987).

Pockets of water above ground, in tree holes or in buttress tanks, are the microhabitats of certain tree anurans. Mention of these elevated microhabitats emphasizes one of the major differences between tadpoles and adult anurans. Clearly, tadpoles do not "find" a place to live nor can they migrate from one microhabitat to another. They develop where the adults put their eggs. In a stream, subject to flooding, some tadpoles will be washed out of their original pool and are unlikely to survive.

2.3.4. Feeding ecology

The anurans of Kalimantan, like those that live in rain forests elsewhere, are generalist feeders. They feed on insects and related invertebrates such as spiders. There are a few dietary specialists among these anurans. For example, the Red-Sided

Sticky Frog (*Kalophrynus pleurostigma*), feeds largely on ants and termites. These modest sized anurans commonly take 50-150 ants or termites at one time (Inger and Stuebing 1989). The Crested Toad (*Bufo divergens*), though feeding on a wider assortment of insects, also takes large numbers of ants and termites. Achmad (1988) reported that stomachs of this toad contained 60% ants and 30% termites and averaged 40 individual prey per stomach. In fact, most of the toads in Kalimantan have the same kind of diet with ants as principal prey. Even the Giant River Toad (*Bufo juxtasper*) follows this pattern, despite its large size. Although it takes slightly larger prey (7.4mm vs. 4 mm) (Inger 1987).

Most anurans in Kalimantan are dietary generalists, though ants, by virtue of their abundance, remain an important type of food for many of them. The small Smooth Guardian Frog (*Rana palavanensis*) eats spiders, beetles, and related insects, but even so, 52% of its prey are ants (Inger and Stuebing 1989). As a rule, the generalists tend to eat large prey and to ingest fewer ants and fewer prey per meal. For example, in five stomachs of the Large-Eyed Litter Frog (*Leptobrachium abotti*), Inger and Marx (1961) found crickets, spiders, and a cockroach having an average length of 1.75 cm; the average size of prey per stomach was 1.5 cm. The Bornean Horned Anuran (*Megophrys nasuta*), which has an impressively wide mouth, eats even larger prey (average about 3.2 cm), tackling and swallowing scorpions as long as 10 cm and snails more than 4 cm in diameter. The biggest prey are taken by Blyth's Anuran (*Rana blythi*) and the Greater Swamp Anuran (*Rana ingeri*), two largest species in Sabah as reported by Inger (1987). Their diets included anurans, crabs, large millipedes, and scorpions.

2.4. Ecology of Amphibia

2.4.1. Variation in adaptation with environmental characteristics

Amphibians, especially the ground dwelling species that have left the water, generally occupy environments that are hostile to their basic physiology. They are ectotherms and have a permeable body covering and therefore they are more susceptible to the stress of the environment than any other tetrapods (Duellman and

Trueb 1994). These animals with combinations of many unique morphological structures, physiological mechanisms, and behavioral responses, have adapted to life in nearly all terrestrial habitats, ranging from the Arctic to the deserts, from elevations of more than 5000m to sea level, even to brackish mangrove swamps (Zug 1993). However, when the diversity of amphibians is examined, physiological limitations are apparent, because the majority of species occupy regions having high ambient moisture and moderate to warm temperatures (Brattstrom 1968).

Colonisation of terrestrial habitats requires amphibians to restrict the inevitable loss of body water while maintaining a moist skin for gas exchange. Their success is best exemplified by the capability of some amphibians to inhabit demanding environments, in the driest parts, where life faces two major physiological problems-scarcity of water and environmental conditions that increase water loss by evaporation.

Water generally makes up 70-80% of the body mass of amphibians; the higher percentages occur for aquatic species (Thorson 1964). Water is exchanged readily with the environment, and water conserving mechanisms could function potentially at several sites. The mechanisms involve: the curtailment of water loss through the skin, modifications of the excretory products of the kidneys, and storage of water in vesicles and tissues (Duellman and Trueb 1994) and modifying behaviour such that they avoid the most inhospitable conditions.

Terrestrial amphibians (with a few exceptions among the anurans) are generally nocturnal, therefore avoiding higher daytime temperatures and lower atmospheric humidity. When diurnally active these animals normally have a higher moisture content than surrounding areas exposed to insolation and air currents. Because of this, the undersides of stones, interior of logs, depths of leaf litter, shaded crevices, tree holes, and axils of leaves of aroids and bromeliads, as well as burrows in the soil, are common diurnal shelters for terrestrial and some arboreal amphibians (Pough 1982). If sufficient water is taken in during the day, the animal can afford to lose water during nocturnal forages. For example, the burrowing Kalimantan horned

frog (*Megophrys nasuta*) experiences an average water loss equivalent to 25% of its body mass while it is foraging each night (Nawangsari 1985). Although many kinds of amphibians may become active during and immediately after heavy rains by day, they are usually active only at night. However, the young of some anurans, particularly bufonids, tend to be diurnal and heliothermic in contrast to conspecific adults. In contrast, some groups of anurans are characteristically active only by day. All these live in habitats where water is readily available. For example, the diurnal *Dendrobates* and *Phyllobates* live in US forested regions having high atmospheric humidity, as do most species of *Bufo* in Kalimantan tropical forests. Other species of *Bufo* that live in areas of lower atmospheric humidity are active only in the surroundings of streams, which they enter frequently (Inger 1987). Other stream-dwelling anurans may be rehydrated continuously from the spray of waterfalls or by entering the streams frequently; these include New World bufonids (*Atelopus*), Asiatic ranids (*Staurois*), and Australian myobatrachids (*Taudactylus*) (Trueb 1971).

Reduction of the amount of surface area exposed to evaporation is another important strategy of reducing water loss. Several elongate plethodontid salamanders curl their bodies and tails into tight coils and therefore reduce evaporative water loss (Ray 1958). Another way to reduce the surface area exposed to the air is by selecting a shaded site and tucking the limbs close to the body, as undertaken by tree frogs. Such frogs characteristically are quiescent; presumably slowing of metabolic processes results in a slower rate of breathing and thus less loss of moisture through respiration (Pough et al. 1983). The behavioral adaptation to prevent water loss in dry habitats is burrowing. Such burrowing behavior is best exemplified by anurans in arid areas. In Arizona, for example, the spadefoot toad, *Scaphiopus hammondii*, spends about 9 continuous months underground in self-made burrows which reach depths of 90 cm (Ruibal *et al.* 1969). By maintaining an osmotic concentration equal to the soil moisture tension, the toads may remain in these burrows for long periods of time without losing water to the soil.

There have been many studies on the rates of water loss and water uptake in amphibians, especially anurans (Ray 1958, Thorson 1964, Alvarado 1967, Lee 1968,

Ruibal *et al.* 1969 and Pough *et al.* 1983) . However, the results of these studies are not necessarily readily comparable because of differences in experimental design. For example, water loss (usually determined as weight loss) has been measured in still air or at relatively rapid rates of convection; also, bladder contents were emptied in some experimental animals and not in others. In experiments concerned with rehydration, the absorptive capabilities of skin on different regions of the body and the moisture tension of the substrate were not always taken into consideration. Nevertheless, some patterns are evident, and some generalisations can be made (Duellman and Trueb 1994).

Unless the skin of amphibians is covered by some special coating essentially "waterproofing" the skin or contains structures that reduce the permeability, the skins of amphibians, irrespective of taxonomic group or habitat, give up water at approximately the same rate. In contrast, rehydration rates are highly variable and depend on structural differences and absorptive properties of the skin; these seem to be related to habitat. Terrestrial and arboreal species have faster rates of rehydration than semiaquatic and aquatic species (Mullen and Alvarado 1976, Brown *et al.* 1977). However, aquatic species can tolerate prolonged hydration better than terrestrial species. Semiaquatic species normally have access to free water and all amphibians have access to abundant moisture during rain. However at times of absence of free water, amphibians must depend primarily on the moisture content of the substrate. For arboreal species the source of moisture may be the condensation on leaves of bushes and trees, but in ground-dwelling species the exchange is with the soil substrate (Hillyard 1976).

Most experiments in relation to dehydration and rehydration have been performed at constant temperatures. Claussen (1969) showed the absence of correlations between rehydration rates and temperature in six species of North American anurans. Although hormonal release and rate of blood circulation increased with temperature, other properties of anurans, especially differential uptake of water by the skin in different regions of the body, seem to negate the effect of temperature.

Significant differences in both dehydration and rehydration rates at different temperature in species of plethodontid salamanders were demonstrated by Spotila (1972). Experimental dehydration was accomplished in desiccators and rehydration culture dishes partially filled with water, both maintained in environmental chambers at 5, 15, and 25°C. Both rates were positively correlated with temperature, and the rehydration rate indicated a greater response to temperature.

A dehydrating amphibian faces a conflict between the need to escape desiccating conditions and the increased cost of such escape in the form of activity with increased evaporative loss. In at least one anuran, *Notaden bennetti* (Heatwole *et al.* 1971), and one salamander, *Plethodon cinereus* (Heatwole 1960), activity (mostly escape behavior or movements along moisture gradients) increases during progressive dehydration until a peak is reached, after which activity declines (burrowing or moisture-conserving postures) until just before death, when a brief, secondary burst of activity (escape movements) occurs.

Amphibians are ectotherms, and generally have body temperatures close to that of their immediate surroundings, especially the substrate. There is no proof of internal heat-production mechanisms that increase the body temperatures of amphibians above that of the environment. The amount of metabolic heat produced that is lost immediately to the environment is small (Fromm 1956). Some anurans bask and thereby raise their body temperatures; however, basking creates problems of evaporative water loss. In warm environments, most amphibian behaviour seems to be related to maintaining low body temperatures, but cryptic and nocturnal activity (in places or at times of lower temperatures) is more likely to be a response to problems of water economy than to temperature (Duellman and Trueb 1994).

Amphibians as a group have a wide range of thermal tolerances; differences in tolerances reflect the different thermal regimes in their habitats. The temperatures at which amphibians are active in the field are known for many species of salamanders and anurans, especially species in North and Central America (Brattstrom 1963, Feder et al. 1982). On the basis of the data on North and Central American

amphibians, the range of temperatures for salamanders to be active for overland travel is -2.0 to 30.0°C (mean = 13.9°C), compared with 3.0 to 35.7°C (mean = 21.7°C) in anurans. In Kalimantan-Indonesia, the range of temperatures for Bornean anurans to be active is 20.0° to 31.0°C (mean = 25.5°C) (Boeadi, pers. comm.).

Complex relationships exist between body temperatures and latitude. For example, Feder and Lynch (1982) found that plethodontid salamanders in the temperate zone experience lower minimum temperatures than neotropical salamanders. The tropical salamanders show similar rates of decline in mean body temperatures with increasing altitude.

Seasonal variation in body temperature is greater among temperate amphibians than among tropical species. However, at a given time and place, variation in body temperatures among members of a population is similar for temperate and tropical amphibians. Species living at high altitudes are subjected to highly variable daily temperatures, whereas those in warm tropical regions have relatively little daily variation in temperature (Duellman and Trueb 1994).

The majority of field data suggest that amphibians seldom maintain a constant body temperature (Huey and Slatkin 1976), but vary with ambient temperatures. The covariance of body temperatures with the environment is evident at the level of individuals, populations and species (Carey 1978). Therefore, most amphibians seem to maintain a constant body temperature from day to day only when the prevailing environmental conditions do not vary. For example, the similar body temperatures of amphibians living in bromeliads in the tropics (Feder 1982, Feder and Lynch 1982) are the result of minimal thermal diversity within the bromeliads. Likewise, the body temperatures of the leptodactylid anuran *Somuncuria somuncurensis* are limited to the nearly constant temperatures (20-22°C) of the thermal springs in which it lives (Cei 1969). Although amphibians seem to avoid extreme temperatures, their body temperatures may be altered by trophic state, acclimatization temperature, developmental stage, environmental moisture, oxygen availability, reproductive state, time of day, and availability of appropriate environmental temperatures (Feder

1982). Of all these factors, moisture probably has the most marked effect. In Spotila's (1972) study, salamanders selected areas of the highest relative humidity in thermal and relative humidity gradients. Thus, there seems to be a definite interplay between thermal and moisture responses.

Body temperatures of amphibians usually are nearer the upper than the lower extremes of temperature tolerated. Brattstorm (1968) recorded the critical thermal maxima for six species of North and Central American anurans to be greater than 40°C, notably higher than the temperatures at which these species are active in nature (Brattstorm 1963); active temperatures were far above the minimum temperatures tolerated. For example, temperatures of 19 active *Smilisca baudinii* were 21.1 to 28.8°C (mean = 24.3°C), whereas the critical thermal maximum and minimum were 40.4 and 5.0°C, respectively. Comparable temperatures for 25 *Bufo marinus* were 22.0 to 27.0°C (mean = 25.2°C), and the critical thermal maximum for salamanders (38°C for *Ambystoma mabeei*; Hutchinson 1961) is 8°C above the highest temperatures recorded in the field for tropical plethodontids (30°C). A similar difference is found in anurans, the highest critical thermal maximum for which is 42.5°C for *Hyla smithii* (Brattstorm 1968).

2.4.2. Variation in diet with prey availability

The feeding strategies of amphibians vary widely both in terms of their choice of prey and methods of prey capture and ingestion. Amphibians are generally considered to be opportunists with their diets reflecting the food availability. This may be true for some, but results of field and laboratory studies show that some species are selective in their feeding (Tyler 1976, Christian 1982, Duellman and Trueb 1994).

Most accounts of amphibian feeding are anecdotal and relate only to a few taxa. Consequently, little is known about prey selection and foraging strategies. The limited information on amphibian diets indicates that all adult amphibians are carnivores; although many species eat a wide variety of invertebrates most feed principally on insects (Table 2.2). Opportunism is apparent in many spp. for

example *Rana hosei* and *R blythi*, most of their diets consisting of invertebrates from moderate to large size. Proportionately more crabs were eaten by *Rana ibanorum* and *R macrodon* than by *R blythi*. This difference may reflect differences in opportunity rather than differences in food preference, as *ibanorum* and *macrodon* tend to sit closer the water's edge where crabs emerge at night than does *blythi*. (Ingers 1969).

Table 2.2. Frequency of taxonomic categories (% occurrence of prey organisms) in anurans' stomachs along rain forest streams in North Kalimantan (Inger 1969).

Prey items	Anuran species						
	Bufo asper	*Pedostibes hosei*	*Rana blythi*	*R. ibanorum*	*R. macrodon*	*R. kuhli*	*Amolops jerboa*
Ants	61	54	35	17	12	27	11
Beetles	6	17	18	9	19	13	5
Orthoptera	-	-	14	17	14	4	21
Termites	29	-	6	29	8	25	16
Lepidoptera	-	-	-	2	19	4	16
Other insects	1	29	4	3	6	7	-
Diplopoda	-	-	5	1	-	4	16
Chilopoda	-	-	6	2	2	2	-
Arachnida	2	-	5	4	6	11	16
Crabs	-	-	2	7	7	-	-
Other invertebrates	-	-	4	8	2	5	5
Anurans	-	-	1	0	5	-	-
Other vertebrates	-	-	1	1	1	-	-

Herbivory is characteristic of anuran larvae, but it may occur in other amphibians; for example, the aquatic salamanders of the genus *Siren* have been reported to have large quantities of vegetable matter in their digestive tracts and to eat the pond weed *Elodea*, as well as aquatic invertebrates (Ultsch 1973). *Bufo marinus* may eat vegetable scraps and other vegetable matter (Alexander 1964, Tyler 1976).

Large anurans, such as *Ceratophrys ornata*, South African bullanuran *Pyxicephalus adspersus* and *Rana catesbeiana* and the Giant River Toad *Rana juxtasper*, commonly feed on large prey items, such as small mammals, birds, turtles, snakes, and other anurans (Inger and Stuebing 1989). Branch (1976) reported a *Pyxicephalus adspersus* that had eaten 17 newly born cobras (*Hemachatus*

haemachatus). Some large salamanders like *Dicamptodon ensatus* also eat plethodontid salamanders, anurans, snakes, mice, and shrews (Bury 1972). But these gastronomic feats are exceptions and general availability of prey of the appropriate size and type seems to be a basic limitation on the diets of amphibians. For example, analyses of stomach contents of three species of ranids (*Rana macrodon*, *R. ibanorum* and *R. blythi*) in North Kalimantan, Indonesia (Inger and Greenberg 1966), five species of salamanders in New England (Burton 1976), *Acris crepitans* in Indiana (Labanick 1976), terrestrial eft stages of *Notophthalmus viridescens* in New York (MacNamara 1977) revealed that the abundance of food items in the stomachs was correlated with the relative prey abundance in the habitat.

2.4.3. Variation in diet with habitat

Individuals of a particular species may exhibit significant differences in the types and amounts of prey eaten in different habitats; as stated above this mainly reflects differences in prey availability among habitats. Inger and Marx (1961) found noticeable differences in stomach contents of several species of anurans at different elevations in the Upemba National Park, Zaire, and Barbault (1974) reported differences in diets of anurans in savanna and forest habitats in the Ivory Coast. In addition, three species of ranids have different diets at different types of lowland forest streams in Sarawak, North Kalimantan as reported by Inger (1969).

Differences on a more local scale also are evident. For example, in freshwater habitats, the diet of *Rana cancrivora* consists mainly of insects, but in nearby brackish water the anurans eat mostly crustaceans (Elliot and Karunakaran 1974). Inger (1989) also reported that *Rana limnocharis* mainly feed on insects in fresh water streams but change their diet to feed more on fish and crustaceans when they live near the river mouth which has brackish water. Newts (*Taricha granulosa*) in a permanent pond eat a greater diversity of prey than do individuals in a temporary pond (White 1977).

2.4.4. Variation in diet with season

Seasonal differences in diets have been reported for various species of amphibians e.g., *Rana pretiosa*, (Turner 1959); *Plethodon glutinosus* and *Plethodon jordani*, (Powers and Tietjen 1974). Studies of diets of many anurans in seasonal tropical environments in West Africa have revealed noticeable differences throughout the year (Inger and Marx 1961). Inger (1969) also mentioned that the diets of several Kalimantan ranids like *Rana kuhli* and *Rana signata* along small forest streams in Sarawak were also different between dry and wet seasons. Among 13 species of anurans dwelling on the forest floor in Amazonian Peru, the diversity of food eaten by some species was greatest in the dry season (Toft 1980).

Seasonal differences in diets reflect availability of prey and, in some cases, seasonal differences in selectivity by amphibians (e.g., certain forest-floor anurans; Toft 1980). This selectivity may be "forced" on the amphibians by factors other than food, in particular by the necessity of foraging under physiologically tolerable moisture conditions. For example, the plethodontid salamander *Desmognathus fuscus* shows a selection for larger prey with increased precipitation (Sites 1978). *Plethodon cinereus* forages in moist leaf litter; when the leaf litter is dry, the salamanders are confined to feeding on limited amounts and kinds of prey occurring under rocks or logs (Jaeger 1980).

The seasonal activity of certain species is determined, in part, by the activity of prey. This is especially evident among prey specialists. The activity of the termite-eating anuran *Breviceps verrucosus* in South Africa is timed to the swarming of termites (Poynton and Pritchard 1976). The period of feeding activity by the spade-foot, *Scaphiopus couchii*, in southwestern North America also is correlated with the swarming of termites (Dimmitt and Ruibal 1980).

Diel activity of prey may account for the predator's feeding activity and, therefore, kinds of prey taken. For instance, peak surface activity of three species of streamside plethodontid salamanders is highly connected with the activity of potential prey at dusk or shortly after dark (Holomuzki 1980). Freed's (1980) analysis of prey

behavior and feeding activity by the tree anuran *Hyla cinerea* suggested that the anurans selected prey in connection to the proportion of time that the prey species was active and the kind of activity displayed by the prey. Therefore, increased frequency of prey activity resulted in a perceived increase in the density of that prey species for the predator, thereby resulting in greater predation. When prey selection was limited to prey types having similar activity patterns, size of the prey species became an important factor in prey selection, with the larger prey being selected (Duellman and Trueb 1994). Analysis of stomach contents of the Bornean ranid *Amolops larutensis*, and activity patterns and abundance of prey species throughout the year led Berry (1966) to conclude that diet selection by this anuran is correlated most closely with activity of the prey.

2.5 Community and Species Diversity

2.5.1. Community structure

Until now theoretical community ecology has tended to assume that interspecific competition is of primary importance in the determination of species composition in most communities (Roughgarden 1983, Schoener 1983). A number of laboratory experiments have demonstrated that interspecific competition can lead to extinction (Gause 1934, Park 1954). Most inferences about interspecific competition concern exploitative competition-use of the same resource by two or more species and indicate interference competition - one organism limiting another species' access to a resource. However, predation may differentially influence the relative abundance of coexisting species and therefore have a significant effect on the result of interspecific competition; also predation may affect the species composition of communities (Duellman and Trueb 1994). Previous re-evaluations of descriptive studies of vertebrate communities (Strong *et al.* 1979) have questioned the statistical validity of much indirect evidence that has been quoted in support of the role of competition in structuring communities. Strong *et al.* (1979) and Simberloff (1983) suggested that much of the purported structural pattern in vertebrate communities simply reflects interspecific variation in ecological attributes usually correlated with

resource utilization, rather than a systematic partitioning of resources in line with competition and niche theory.

Studies of amphibian communities have been uneven in their approaches and thoroughness. Most studies of salamanders have concentrated on the ecological relationships of two or just a few sympatric or parapatric species, whereas many anuran communities have been studied but in less detail. Nothing is known about caecilian communities (Duellman and Trueb 1994).

It is important to contrast between experimental and observational studies of amphibian communities. In experimental studies (Inger & Greenberg 1966, Hairston 1981, Morin 1981) the numbers of one or two species are directly altered, and changes in the abundance or behaviour of other species are monitored. Such experiments can indicate unequivocally the existence of interspecific interactions. However, without a detailed knowledge of natural history, the mechanistic basis of the interaction may be obscure (Hairston 1981), therefore making it difficult to evaluate the generality of the experimental results. Furthermore, for significant logistical reasons, experimental manipulations usually involve only a few species, may extend through only a fraction of a generation, and may be held at spatial scales inappropriate for understanding community dynamics.

In contrast, most observational studies emphasize resource partitioning - the differential utilization of the physical and/or biotic environment by different species. Generally this has been achieved by measurements of a resource matrix (niche breadth) of a species and of the association of two or more species with regard to one or more resources (niche overlap) (Hurlbert 1978).

A comparison of anuran communities in adjacent areas of broadleaf evergreen forest, deciduous dipterocarp forest, and agricultural land in northeastern Thailand (Inger and Colwell 1977) revealed that in seasonally dry regions only 24 species of anurans occurred. Nineteen species were found in the evergreen forest, 20 in deciduous forest, and 19 in the agricultural land, with the greatest faunal overlap between

evergreen and deciduous forests. They suggested that unpredictable environments tend to prevent the formation of distinct guilds, which are an expression of specialization in resource use; therefore the greater species richness of more predictable habitats may be a function of guild formation.

Interspecific competition is often advanced as an explanation for replacement of species by ecological counterparts. Where environments are stable, as in controlled laboratory experiments, interspecific competition usually leads to the stress or extinction of one species. In a complex natural environment, fluctuations in physical and biotic factors may long delay competitive exclusion (Inger and Greenberg 1966).

2.5.2. Species diversity

The concept of species diversity as a unitary measure involving both the number of species (richness) and relative numbers of individuals per species (equitability) has received considerable attention (Pianka 1977). Species richness is a count of the number of species in the community, whereas equitability indicates how individuals in a community are distributed among species (the eveness of species abundance) (Stuebing 1992). It is dependent upon the number of species present and the sample size (Hurlbert 1971, McNaughton and Wolf 1973). The theoretical implications of differing degrees of equitability are not clear, but they are thought to be related to community maturity, and the predictability and stability of the environment (Lloyd *et al.* 1968, Emlen 1973). If all species in a community contain the same number of individuals, the apportionment is maximally equitable. If some species are abundant and some rare, the distribution is inequitable.

It is commonly known that the number of species of ectothermic vertebrates like anurans is higher in the tropics than at high latitudes (Table 2.3). Accordingly, broad patterns of species richness in amphibians as a group show latitudinal trends

Table 2.3. Latitudinal Gradient in Anuran Species Richness in the New World

Site	Degree N. latitude	Number of species	Reference
George Reserve, Michigan	42	8	Collins (1975)
University of Kansas, Reservation	39	9	Fitch (1965)
Brazos County, Texas	31	11	Wiest (1982)
Tehuantepec, Mexico	16	17	Duellman (1960)
Barro Colorado Island, Panama	9	19	Myers&Rand (1969)
Santa Cecilia, Ecuador	0	81	Duellman (1978)

However, integrated with the latitudinal trend are trends along moisture gradients. For example, in North America, generally high numbers of species occur in areas of high rainfall, primarily in southeastern United States and secondarily in the northwest.

Comparisons of amphibian species richness among various regions in the tropics stress the importance of moisture in determining the richness of the amphibian fauna. For example, a strong decline in the number of species from south to north in the Yukatan Peninsula is closely correlated with the amount of rainfall (Lee 1980). Heyer (1967) shows an increase in the total number of amphibian species from 46 in the dry Pacific-slope lowlands of Costa Rica to 81 species in the adjacent wet upland areas. A latitudinal gradient in relation to decreasing moisture is evident in the numbers of species of amphibians in the New World; the moist Paramos in Equador support up to five species, whereas only one species is present in the dry parts of the Altiplano in northern Argentina (Pefaur and Duellman 1980). Even at the same latitude, the effect of this moisture gradient is apparent. At Belem, Brazil, at the mouth of the Rio Amazonas, rainfall amounts to about 2800 mm annually but is evenly distributed (most of the rain falls in one 6-month season); 37 species of anurans are known from the vicinity of Belem (Crump 1971). In contrast, at Santa Cecilia, Ecuador, in the western part of the Amazon Basin, around 4400 mm of rain falls throughout the year; 81 species of anurans are known (Duellman 1978).

The effects of seasonal periods of dryness are less simple. Within anuran communities inhabiting leaf litter in tropical forests, species richness is positively associated with the amount of rainfall and negatively associated with the number of dry months (Wiest 1982). Increasing seasonality as measured by the number of dry months, reduces the number of species present, with about four amphibian species dropping out for each additional dry month in Costa Rican tropical forests (Scott 1976). The reduction in anuran species is certainly a result of the absence of permanent water supplies, complete lack of tank bromeliads and small, fast-flowing streams and a general dry-season scarcity of moist microhabitats. Amphibians like dendrobatid and small eleutherodactyline anurans are very active and probably have high energy and moisture demands. Thus, seasonality in water sources can be seen to affect anuran faunas in the same way that seasonality in insect or fruit supplies affects birds (Heyer and Berven 1973).

2.5.3. Abundance and equitability

Only few quantitative studies have been carried out to allow evaluation of the abundance of amphibians in different regions, although some individual species, such as the salamander *Plethodon cinereus*, have been studied in detail (Duellman and Trueb 1994).

The only detailed comparisons of amphibian abundance in tropical forests are the works of Lloyd *et al.* (1968) and Inger (1979) in southeastern Asia, and the analyses of forest floor communities in Central America, Africa and southeastern Asia (Scott 1976, 1982; Inger 1979). These comparisons indicate that both terrestrial and arboreal anurans were far more abundant in diurnal and nocturnal samples from forests in northern Kalimantan and peninsular Malaya than in seasonally dry forests in northeastern Thailand. For example, the mean number of terrestrial, nonriparian anurans captured per day at Sakaerat, Thailand, was 0.12 in dry evergreen forest and 0.27 in deciduous forest, as compared with 1.31 per day at Nanga Tengalit in Kalimantan. However, these figures are extremely low in comparison with those from Central America - 11.6 for Osa, Costa Rica, 14.7 for La Selva, Costa Rica, and 29.8 for Silugandi Panama. The difference in abundance of anurans is related to

differences between the local climate in Kalimantan, on the one hand, and north-eastern Thailand (Sakaerat) on the other. The shifts are, therefore, associated with a major change in climate. The Thai forests sampled lie in a region having a pronounced dry season and many days with high temperature (>35°C) and low relative humidity (<50%). This relatively harsh climate is particularly unfavourable for anurans. Besides the evidence from numbers of individuals, this effect can be seen in the conspicuous difference in numbers of species of anurans in Kalimantan (total species 33-52) and in the Thai environment (19-20) (Inger 1979). The response of the anurans to the change in climate is understandable as these animals are generally susceptible to desiccation at high temperatures. In Sakaerat, Thailand, the species occurring in both forests were more frequently found under cover in the deciduous forest, which has the harsher climate (Inger and Colwell 1977).

The most striking differences in amphibian abundances are those occurring for forest dwelling species/forest litter anurans between Costa Rica and Panama, and in comparable situations in Kalimantan. The lowland habitats of both forests are probably structurally similar. A comparison between mean sizes of the largest trees in each plot in Kalimantan and pooled Osa, Costa Rica samples showed no significant difference. The litter in the two areas appears to be similar based on qualitative descriptions for Kalimantan (Inger, pers.comm.) and quantitative measurements in Costa Rica. However abundances of amphibians are far greater in Costa Rica than in Kalimantan.

Divergent densities may be a result of either basic differences in the composition of the litter animal community or differences in the functioning of the total system. Compositional differences might be reflected in either the existence of more predators or more non-herpetological competitors. Snakes probably take more litter anurans than any other group of predators. Lowland snake densities determined from the plots reported here are positively correlated with densities of the herpetofauna (Costa Rica 0.22, Kalimantan 0.09 snakes/100m^2), a condition suggesting that predation is probably not the reason for the dramatic differences. The importance of non-herpetological competitors is impossible to estimate with the available data.

However, the scant anecdotal information suggests that none of the competitors are abundant enough to explain the tenfold differencies in lowland densities (Scott 1976).

If there are no important compositional differences in the forest litter communities, perhaps there are fundamental functional differences in the forest ecosystem. The kind of difference in abundance observed here may be characteristic of all Asian-New World faunal comparisons. Janzen (1974) has commented on the extreme scarcity of animals of all kinds in Bornean forests, and the scarcity of small litter vertebrates probably reflects this generality. Teams of collectors were only able to amass an average 24.1 amphibians per day on two Kalimantan sites with good soil (R. Inger pers.comm.) compared with the usual 50-100 specimens a day collected by a single collector in Costa Rica and Panama (Janzen 1974). Bray and Gorham (1964) summarized the available worldwide litter-fall data. Their data show that two Malayan dipterocarp forests produced an average of 6.4 metric tonnes/ha/yr of litter in contrast to an average of 12.2 tonnes produced in five different American and African lowland wet tropical forests. Janzen (1974) suggests that wet tropical forests on nutrient-poor soils can be structurally similar to forests on better soils, but that plant growth and turnover rates are much slower. If this supposition is true, then litter fall is less, decomposition slower, and the animals dependent upon the litter energy may be much more scarce. Therefore, the abundance of anurans in Kalimantan lowland tropical forests is much lower than in the Costa Rican forests

It has been well documented that in many groups of organisms, species richness is high in tropical forests, but a fallacy that has persisted in the literature is that in tropical forests no one species is common, thereby implying that equitability is high. In addition, Maiorana (1976) stated there are many species in the tropics, but generally individuals of any one of them are rare. This may be true for some organisms, such as trees, but considerable evidence indicates that this is not true for amphibians. For example, in large samples of anurans inhabiting the leaf litter in tropical forests in Kalimantan, the Philippines, and Central America (reviewed by Scott 1976), the most abundant species was represented by at least twice the number

of individuals of the second-most abundant species, and in some cases the most abundant species was represented by more individuals than all the other species combined. In an aseasonal tropical rainforest at Santa Cecilia, Ecuador, 5665 anurans representing 81 species were collected (Duellman 1978). The 5 most abundant species represented 22% of the total number of individuals, whereas the five least abundant species (4 with one individual each, and 1 with two) represented only 0.1% of the total.

Comparable data are available from Lost Lake, South Carolina, in the eastern United States (Bennett *et al.* 1980). Among five species of salamanders, sampled over 2 years, the most abundant *Notophthalmus viridescens* composed 85% and the least abundant *Ambystoma opacum* only 0.1% of 3641 individuals. Among 11 species of anurans, *Gastrophryne carolinensis* (36%) and *Bufo terrestris* (29%) were the most abundant, in a sample of 11,381 individuals, whereas 2 species, *Hyla squirella* and *H. versicolor* the least abundant, were each represented by only a single individual. The figures for temperate zone anurans are only slightly different from those for tropical environments.

The decline in species richness, but increase in abundance of individuals with increasing altitude, results in lower equitability at higher elevations. This was shown by Scott's (1976) analysis of litter plots at different elevations in Costa Rica (Table 2.4), which indicates a twofold difference in relative abundance of the most common species at a mountain site (San Vito) as compared with lowland sites (La Selva and Osa).

Table 2.4. Equitability of Anurans in Leaf Litter at Different Elevation in Costa Rica (Scott 1976)

Locality	Elevation (m)	Number of plots	Number of species	Number of individuals	Most abundant (%)	Second-most abundant (%)
La Selva	100	19	15	165	42	19
O s a	20	20	24	135	37	18
San Vito	1200	10	8	266	83	6

This comparison is especially meaningful between one lowland site, Osa, and the montane site, San Vito, because the same species, *Eleutherodactylus stejnegerisnus*, is the most abundant anuran at both sites (Scott 1976).

2.6. The Global Status of Amphibians

Historically, amphibians have not received the conservation concern accorded groups such as birds and large mammals. This may be due, in part, to less popular interest in amphibians, but it is also due to poor knowledge of population processes and trends for most amphibian species (Drost and Fellers 1996). In the last few years there has been increased concern about amphibians following reports of population declines in frog and toad species in many areas of the world (Brainaga 1990, Blaustein and Wake 1990, Wyman 1990, Wake 1991, Tyler 1991). That declines were occurring at an alarming rate in widely separated areas of the world also served to draw public attention to the issue (Peachman *et al.* 1991).

Recent dramatic and unprecedented losses of amphibian populations, and even entire species, have been reported from parts of Europe, Asia, Australia, North America, and South America (Barinaga 1990, Blaustein and Wake 1990). These losses are of particular concern because they are occurring on a world-wide basis and because they include species inhabiting large parks and wilderness areas which are not subject to obvious habitat loss. However, many declines in the world are clearly linked to habitat loss or degradation (Vial & Saylor 1993). Other losses do not have obvious explanations, but some tentative patterns have emerged: 1) reported declines have been mainly or entirely among anuran species (frogs and toads); 2) certain taxa and geographic areas seem to be more affected than others (in western North America, for example, most reports have concerned anurans of the genera *Rana* and *Bufo* (Hayes & Jennings 1986, Vial & Saylor 1993); 3) many of the reported declines have been among high-elevation species (Wake 1991); 4) some declines and losses have been among species with restricted geographic and habitat ranges or among populations of more widely distributed species at the edge of their

range (Feller & Drost 1993, Pounds & Crump 1994); and 5) the declines appear to be recent, with well-documented declines and local or regional disappearance of species within the last 20 years (Carey 1993, Bradford *et al.* 1994, Pounds and Crump 1994).

Several western North American anurans are among those that have suffered serious, unexplained losses. Two species with restricted ranges, *Bufo hemiophrys baxteri* (the Wyoming toad) and *Rana onca* (relict leopard frog), are close to extinction (Drost and Fellers 1993). Formerly widespread and abundant species have also been affected: dramatic losses have occurred in *B. boreas* (the western toad) in the Rocky Mountains (Carey 1993, Corn and Bury 1993) and repeated reproductive failures have been reported in the Pacific Northwest (Blaustein & Olson 1991, Blaustein *et al.* 1994). Populations of *R. muscosa* (the mountain yellow-legged frog) have disappeared from over 75% of study sites they formerly inhabited in the mountains of California (Bradford *et al.* 1994). *Rana aurora* (the red-legged frog) has disappeared or declined over much of its range in California (Moyle 1973, Hayes & Jennings 1986), and declines of *R. pipiens* (the northern leopard frog) have been reported across the species' range in the western U.S. and Canada (Clarkson & Rorabaugh 1989, Hayes and Jennings 1986, Bishop and Petit 1992). The cascades frog (*Rana cascadae*) in the vicinity of Lassen Volcanic National Park, California, USA has suffered serious declines. These declines seem to have been caused by a combination of local factors, including (1) the presence of non-native, predatory fish which have restricted habitat and limited dispersal of anurans; (2) loss of breeding habitat due to a five-year drought; and (3) the gradual loss of open meadows and associated aquatic habitats. The loss of anurans suggests that some common management practices in parks and wilderness areas may have endangered some of the species these areas are intended to protect (Fellers and Drost 1993). Similar local factors may account for amphibian declines reported elsewhere and should be carefully evaluated along with possible global effects.

The alteration of habitat is one of the major threats to amphibian populations (Blaustein and Wake 1990). For example, *Nyctibatrachus major*, a rare endemic

anuran in the Western Ghats, India has recently suffered as a result becoming more rare. Recently, denudation of the forests for timber, fuel and collection of material to furnish organic agricultural mulch has greatly affected the habitat of this anuran. A resulting decrease in canopy cover has allowed more direct light penetration to the exposed streams and their margins. Moreover, forestry and mulch collection have increased soil erosion and consequently the silt-load of formerly pristine forest streams and the distribution of *N. major* is decreasing every year as more habitat becomes affected (Krishnamurthy 1997).

The European tree frog (*Hyla arborea*) populations in southern Sweden have been reported to be in decline and their rate of decline is accelerating (Edenham 1996). Reductions in the species' range and a decrease in numbers in Sweden have been noted throughout this century until the mid 1990s with 14-15% of the populations becoming extinct.

The drastic decline of anurans and salamanders in different regions of the world has aroused the scientific community to action. As a consequence, the International Union for the Conservation of Nature/Species Survival Commission (IUCN/SSC) established the Declining Amphibian Populations Task Force (DAPTF) in 1992 for the purpose of: (1) Facilitating the Organisation of a global network of Working Groups to monitor and assess the status of amphibian populations, (2) Promoting studies on the effects of potential causes of observed declines, including consideration of indirect and synergistic factors and (3) Recommending uniform protocols and providing a world-wide information network on the status of amphibian declines and ongoing monitoring activities.

The question is frequently raised as to how long it will be before the Task Force can give accurate information about the global status of amphibian declines. There may be geographic areas where information will be difficult to obtain for many more years. Experimental studies examining potential causes of amphibian declines may raise questions as well as provide answers. It is therefore a difficult task and a challenge as a Task Force to determine when they have achieved a critical body of

data that will provide a scientifically credible answer as to whether observed amphibian declines are caused by regional or global factors. The necessary body of evidence about amphibian declines is now emerging in the scientific literature, but it has not reached the stage of critical mass. Amphibian declines are a complex and multifaceted problem which defy simple explanations and need long-term and thorough studies to give a scientifically valid answer.

Table 2.1 Checklist of Frogs and Toads reported from Kalimantan (Inger and Stuebing 1989)

Latin Name (Collector)
Family Pelobatidae
Leptobrachella baluensis Smith
L. parva Dring
Leptobrachium montanum (Fischer)
*L. abotti**
Leptolalax gracilis (Guenther)
Megophrys baluensis (Boulenger)
M. edwardinae
*M. nasuta**
Family Bufonidae
Bufo asper Gravenhorst*
B. juxtasper Inger*
B. *divergens* Peter*
Ansonia albomaculata Inger
A. fuliginea (Mocquard)
A. guibei Inger
A. hanitschi Inger
A. leptopus (Guenther)*
A. longidigita Inger*
A. platysoma Inger
A spinulifer (Mocquard)*
Pelophryne misera (Mocquard)
Leptophryne borbonica (Tschudi)*
Pedostibes everetti (Boulenger)
P hosei (Boulenger)*
P maculatus (Mocquard)
P rugosus Inger*
Family Ranidae
Rana baramica Boettger
R blythi Boulenger*
R cancrivora Gravenhorst
R erythraea (Schlegel)*
R finchi Inger*
R glandulosa Boulenger
R hosei Boulenger*
R ingeri Kiew
R kuhli Dumeril & Bibron*
R limnocharis Boie*
R luctuosa (Peters)
R nicobariensis (Stoliczka)
R palavensis Boulenger*
R paramacrodon Inger*
R rugulosa Wiegmann
R signata (Guenther)*
*R. asperata***
*R rhacoda***
Micrixalus baluensis (Boulenger)

Table .....contd.

Latin Name (Collector)

Occidozyga baluensis (Boulenger)
O laevis (Guenther)
Staurois latopalmatus (Boulenger)
S natator (Guenther)*
S tuberrilinguis Boulenger
Amolops amoropalamus Matsui
A cavitympanum (Boulenger)
A jerboa (Guenther)*
A kina baluensis Inger
A orphnocnemis Matsui
A whiteheadi (Boulenger)

Family Rhacophoridae
Philautus amoenus Smith
P aurantium Inger
P disgregus
P ingeri Dring
P longicrus (Boulenger)
P mjobergi Smith
P petersi (Boulenger)
P tectus Dring
Polypedates colletti (Boulenger)
P leucomystax (Boie)*
P macrotis (Boulenger)*
P othilophus (Boulenger)*
Rhacophorus acutirostris Mocquard
R appendiculatus (Guenther)*
R baluensis Inger
R bimaculatus (Peters)
R dulitensis Boulenger
R everetti Boulenger
R gauni (Inger)
R kajau Dring
R nigroplamatus Boulenger
R pardalis Guenther
R reinwardti
R rufipes Inger*
Nyctixalus pictus (Peters)

* obtained during night riparian transect
** New species for Kalimantan (Collected by Agustinus W Taufik, January 1995) at Kalang River, Central Kalimantan, Indonesia

CHAPTER III.

LOGGING AT THE PROJECT SITE (CENTRAL KALIMANTAN-INDONESIA)

3.1 Introduction

Indonesia has 10% of the world's rain forest and 40 to 50% of the rain forests of Asia. These forests contain approximately 4000 tree species, approximately 500 species of mammals (including 100 endemic species) and 1500 species of birds (representing 17% of the worlds afivauna) (Sormin 1992). The total area of forests is registered as 144 million hectares: 113.3 million hectares of permanent forests (protection 30.3 million ha, conservation/fully protected 18.7 million ha and production 64.3 million ha) and 30.7 million hectares as forests for conversion to agriculture (Ministry of Forestry 1991).

Although Indonesia possesses 144 million ha of legally recognised "forest land", approximately 20 million ha of this area is not forested and less than 13% has been legally conserved as protected areas. The conservation status of the protected areas however is not satisfactory and many areas are threatened with conversion to alternative land uses such as agriculture and plantations. Similarly, the production forests of Indonesia, which total 60.4 million ha, have not been optimally managed for timber and wildlife. The original concept of the Government of Indonesia whereby concessionaires would act as responsible stewards of the forests has not proved effective. Shortages of Ministry of Forestry field staff and lack of motivation on the part of concessionaires are largely to blame.

Forestry in Indonesia is approaching a critical stage in its development. Timber harvesting is proceeding apace, often with political rather than environmental controls, and within a few decades will probably affect most remaining pristine rain

forest areas. Yet the harvesting (logging) systems employed, even where long-lease (contract) agreements are in force and active management applied, seem unsustainable (Ministry of Forestry 1995).

The survival of Indonesian rain forest wildlife crucially depends on the success with which forest managers prevent further reduction in the area of forest estate and the ability of conservationists to maintain and implement protected areas status. This will require wildlife ecologists and foresters/sylviculturists to work together to develop techniques for limiting the adverse effects of logging on wildlife, and, ultimately, for determining systems of harvesting that are economically and biologically sustainable. Some initiatives have been taken for example, the Indonesian Government has developed a selective cutting with replanting (TPTI) system designed to achieve sustainable harvesting of forest in Indonesia. In addition, each forestry logging concession now has to conduct an Environmental Impact Assessment (EIA) and present an Environmental Impact Statement (EIS) prior to logging to the Ministry of Forestry (Ministry of Forestry 1987). But much more needs to be done at the political and practical level if the valuable wildlife assests of Indonesian forest are to survive.

Within Indonesia, the Kalimantan tropical rain forests are some of the most diverse forests. The Indonesian Ministry of Forestry (1993) recorded more than 200 species of dipterocarps (the most valuable commercial timber trees in Southeast Asia) and an estimated 10,000 flowering plants inhabit the Kalimantan forest. Indonesia has an impressively high animal species richness, including mammals (515 species, 36 percent of which are endemic), swallowtail butterflies (121 species, 44 percent endemic), reptiles (600+ species), birds (1519 species, 28 percent endemic), and amphibians (270 species of which around 130 species are found in Kalimantan, Ministry of National Development Planning 1993).

3.2. Logging Management and Regulations

3.2.1. Management

In 1966 the Government of Indonesia (GOI) decreed that Production Forests should be developed by private companies operating under government supervision. In 1967, the government opened up the outer islands (Sumatra and Kalimantan) to timber concessionaires and currently there are 576 concessions covering over 60 million ha of Indonesia (c. 41% of total forest land)(Ministry of Forestry, 1996).

The concessionaire system gives loggers 20-year harvesting rights over a specific area of forest land. Concession holders are legally obliged to regenerate and care for the forest. Regulations require that the processing and marketing of timber be conducted in accordance with a plan agreed upon with the Ministry of Forestry. Management of the concession area should be carried out under three development strategies: a 20-year overall work plan, 5-year work plans and annual work plans. These plans theoretically incorporate conservation concerns and utilise sustained yield principles on a short and long term basis (Director General of Forest Utilization 1987).

Relatively modest changes in the ways in which concessionaires manage forests and logging operations could contribute enormously to increasing the value of production forests for the conservation of biological diversity. In recognition of this some steps have already been taken to achieve the sustainability of the Indonesian forest (Ministry of Forestry 1995):

1. Codes of practice for logging and silviculture developed specifically for Indonesia but based on the ITTO guidelines for sustainable forest management have been adopted.
2. The management of protected areas and production forests must be integrated in an overall land-use management plan.
3. Multiple-use forests have been established following some good examples found in Java.

4. Regulations requiring environmental impact assessments and the establishment of buffer zones have been enforced for each logging concession since 1984.
5. The need for research, extension and training in biological diversity conservation has been recognised and promoted.

Presently, the GOI have been encouraging several forest research institutions by supplying additional funding to focus on forest biodiversity research concerning the ecology of forest flora and fauna, their status and the possible impact of logging. Moreover, the biodiversity implications of forest production areas have recently received more attention from the Indonesian Ministry of Forestry. Consequently there has been an increasing expectation that forest production, in which the primary focus is commodity production, will serve an active role in biological conservation, without compromising its potential for economic productivity.

3.2.2. Regulations

There are two logging systems currently in operation:
1) Clearcutting : This consists of clearcutting with artificial regeneration (modified for mangrove forests). This system is widely used in plantation development (pine *Pinus merkusii* and teak *Tectona grandis*) in Java
2) Indonesian Selective Cutting and Replanting (TPTI): Designed for dipterocarp forests with minimum stem cutting diameter of 50 cm at breast height (DBH).

In addition to prescribed silvicultural systems, concessionaires are expected to adhere to specified restrictions when logging in areas such as: (i) the edge of water springs; (ii) within 50m to 200m buffer zones along river courses (according to the size of the river); (iii) in buffer zones around lakes and reservoir, and (iv) near the edges of steep land.

In 1979, the Director of Nature Conservation launched an initiative that required 1-2% of concession areas to be set aside for wildlife refuges and sanctuaries. This initiative was given added legal force by a decree of the Minister of Agriculture in

1981 which attempted to integrate biological conservation objectives into production forest by setting aside areas within concessions where logging would be prohibited. This very laudable decree was never enforced because it was not considered an urgent priority. The conservation of biological diversity was to be assured by the designation of 49 million hectares as Conservation and Protection forests. The concept did not perish however and in 1989, the Director General of Forest Utilisation ordered all concessions to set aside 100 ha of seed stands for each 5-year plan for cutting areas. These forest stands are to be selected from the least disturbed virgin forest and must be fenced and signed accordingly.

The Ministry of Forestry announced another potentially constructive decree in 1989 which made an advance environmental impact assessment (EIA) mandatory for all existing and proposed timber concessions and industrial plantations. The coordination of EIA implementation is the responsibility of the Director General of Forest Protection and Nature Conservation (PHPA). By 1990, the Director General of PHPA was evaluating EIAs for ten concessions with forty three additional EIAs being prepared. In 1990, The Director General of Forest Utilization of the Ministry of Forestry announced another valuable measure for concessionaires to leave unlogged buffer forests 500m wide adjacent to marked boundaries of protected areas and 1000 m buffers for unmarked boundaries.

3.3. The UK- Indonesia Tropical Forest Management Project (ITFMP)

The ITFM Project (established in 1992) is a part of a collaborative project between UK Overseas Development Administration (ODA) and the Indonesian Ministry of Forestry. The Edinburgh Centre for Tropical Forests (ECTF) provides technical assistance to the Indonesian Forest Research Institute (BPK), Central Kalimantan throughout the four year-project period. The research component of the programme involves scientists and forestry managers from BPK, Kayu Mas Forestry Logging Concession and ODA (Overseas Development Administration) co-ordinated by ECTF.

The research project aims to develop and improve methods for the sustainable management of Dipterocarp forests in Kalimantan. The immediate outputs from the project are designed to support forest management and particularly the development of a Unit Management for the sustainability of Forest Production (KPHP). According to the Indonesian Ministry of Forestry Decree No. 200/KPTS-II/1991, KPHP is defined as the smallest unit of production forest, which can be managed in an economically optimal and sustainable way. The project will produce models for growth and yield prediction and recommend strategies to reduce damage to the forest ecosystem resulting from logging.

The general objectives of the ITFMP (Anonymous 1995) are:

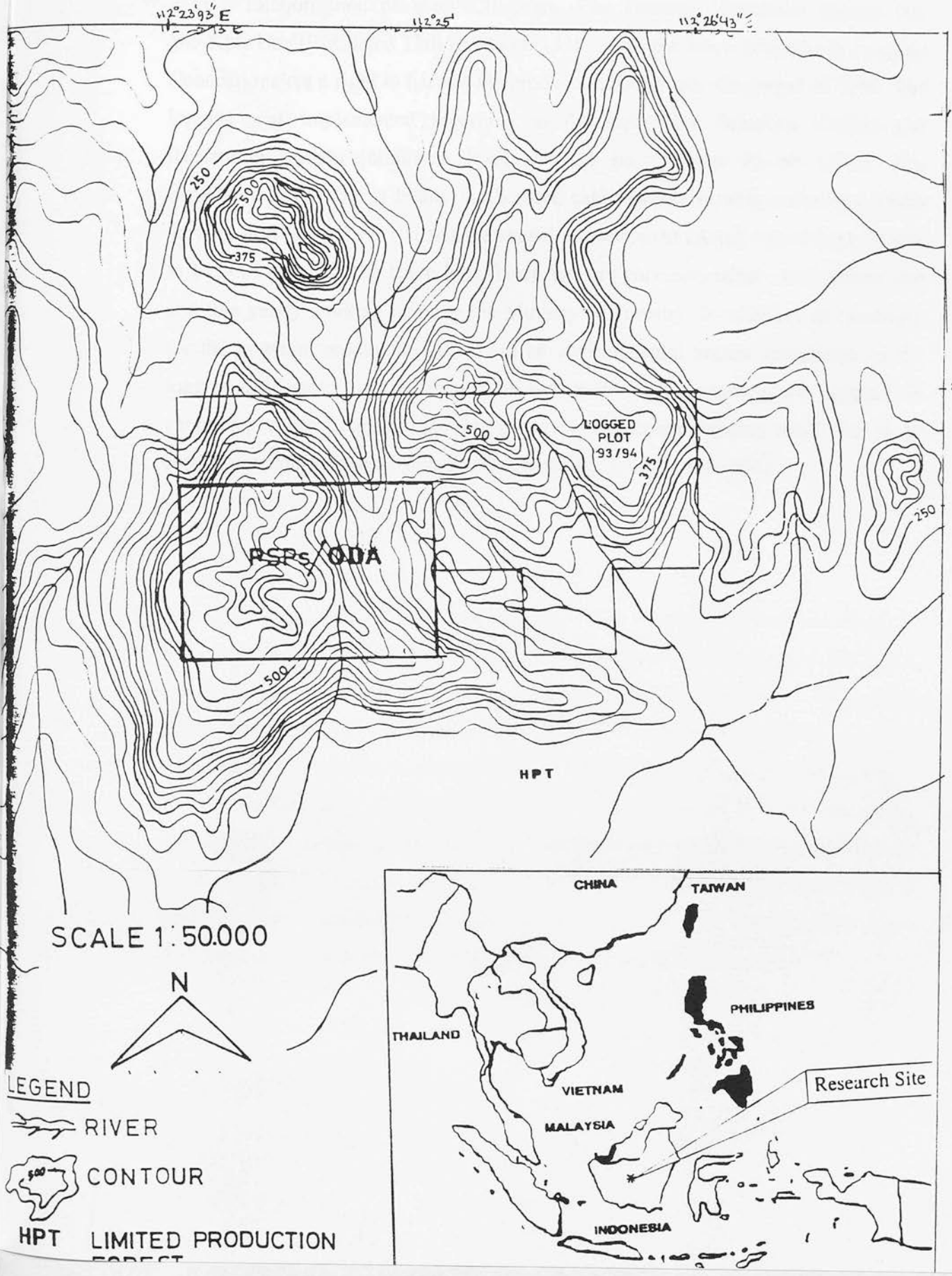
1. To develop a system for yield estimation in managed lowland hill dipterocarp forest that can be integrated with the Indonesian Selective Cutting and Replanting (TPTI) concept.
2. To assist in developing methods for reducing losses in yield to logging damage and develop computer models to improve the prediction of growth and yield estimates for managed dipterocarp forests.
3. To develop practical silvicultural techniques for enhancing the composition of higher value dipterocarp species in forest regenerating after logging.
4. To provide an essential reference text for foresters and forest researchers: a Tree Flora of Indonesia including the major families of economic importance.

The UK-Indonesia TFM research project is located at Wanariset Sangai at Camp 92 within the Kayu Mas Forestry Logging Concession in Central Kalimantan (Map 1).

3.4. Kayu Mas Forestry Logging Concession

The logging concession is located at Camp Tumbang Manyak (Long.1°29'S, Lat.112°31'E) 48 km by road north of the town of Sangai, Central Kalimantan. It lies at the head waters of the Sampit River, about 150 km NNW of Sampit town. The logging concession falls within the administrative responsibility of Kota Waringin Regency and under the supervision of Central Kalimantan Forestry Service. The

Map 1. The Location of the Research Sites (Selective Logged Plots and the Permanent Sample Plots) within the Kayu Mas Forestry Logging Concession



concession covers approximately 87,000 ha of lowland dipterocarp forest on hills reaching elevations of about 300 m.

The silviculture technique applied in the logging concession was a monocyclic system with a rotation time of about 20 years. The Forestry Ministerial Decree No. 145/Kpts/Um/1978 dated 13th of March 1978, gives the Kayu Mas Group Logging Concessionaires a right to harvest the production forest until the period of 1998. The logging was implemented according to the Indonesian Selective Cutting and Replanting System (minimum legal size for harvest was 50 cm DBH). The exploitation of timber is highly mechanized capital-intensive with extraction levels varying from 10 to 18 m^2/ha and the annual allowable cut (AAC) varied from 91,000 to 145,000 m^3/year. The Kayu Mas Group forestry concessionaires must submit one and five yearly working plans to the Ministry of Forestry. In addition, an inventory for the potential production forest and an environmental impact assessment of the logging concession must be carried out before the logging starts. Once logging is complete, a post logging inventory for the remaining standing stocks has to be carried out, and followed by enrichment planting (Anonymous 1994).

3.5. Study sites

Permanent Sample Plots (PSPs)

The site selected for field study was an area of hilly dipterocarp rain forest along the Kalang River at Camp 92 (about 44 km from Wanariset Sangai) within Kayu Mas forestry concession in Central Kalimantan. The study sites were located in the PSPs and within the logged forest areas of Kayu Mas. The PSPs were selected as being representative of pristine forest characteristic of large areas of typical dipterocarps in Kalimantan and very rich in animal and plant diversity. One of the PSPs plot (1 ha) contained 200 species (Proud 1995), greater than Kartawinata (1978) reported for Kalimantan lowland rain forest (138-180 species per ha of forest). The forest stand in the PSPs was dominated by Dipterocarp species (around 57% meranti, *Shorea sp.*) with large emergent legumes such as *Parkia*, *Koompassia*, and *Intsia*.

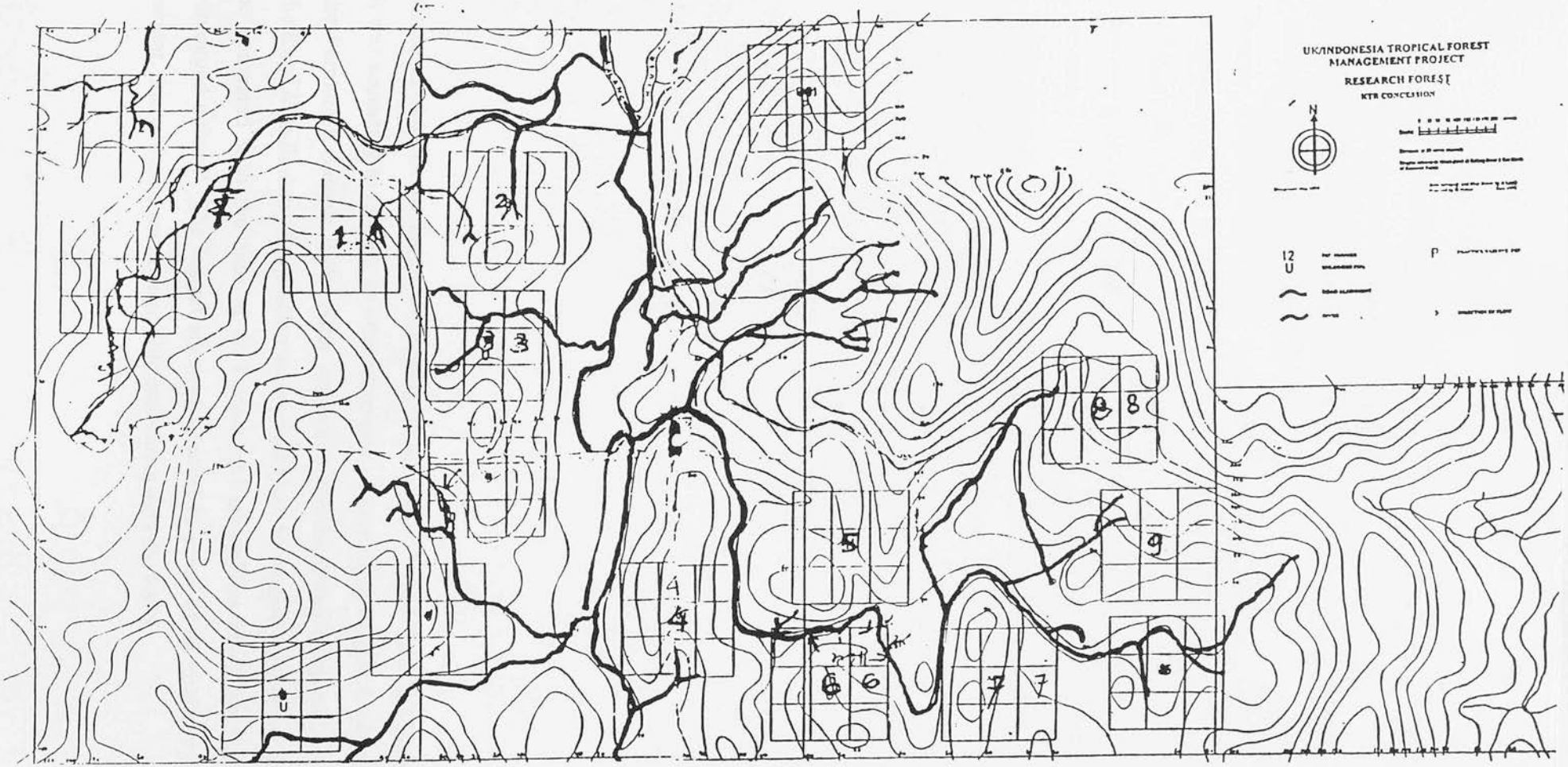
The climate of this area is determined primarily by East and West monsoons and by the movement of the intertropical convergence zone (Anonymous 1993). The climate of rain forest in this study area is typified by having significant dry season, an average temperature of c. 25°C and a fairly constant relative humidity between 85-95% (see Chapter V). Rainfall, however, varies considerably and annual rainfall ranges from 2,800 to 3,900 mm/year.

The limited data concerning geology of the area indicate the area is of granite overlain with Utisols (United States Department of Agriculture Classification) or Acrisols (FAO classification) (Proctor 1994). The top soil has a thin (c. 5 cm) brown horizon with a weak granular structure and many roots and signs of biological activity. Yellowish horizons below these are increasingly firm.

Granulometric analysis revealed mean silt: clay ratios of surface layers of 2.03 (n = 15 samples, range 1.38 - 3.08). A silt: clay ratio of >2 indicates high erodibility (Touber at al. 1989) and this combined with surface layers that were friable and overlying massive resistant lower horizons renders soils in the PSPs highly susceptible to erosion.

Most of the study areas (> 40%) are hilly composed of steep slopes and rugged terrain. The map of the 600 ha research area is flawed in that it shows the slope to be less steep than it actually is. Proctor (1995) reported that the PSPs were on rugged terrain, most of the soils had a surface horizon a few cm thick and there was an increasing risk of soil erosion with increasing disturbance and slope.

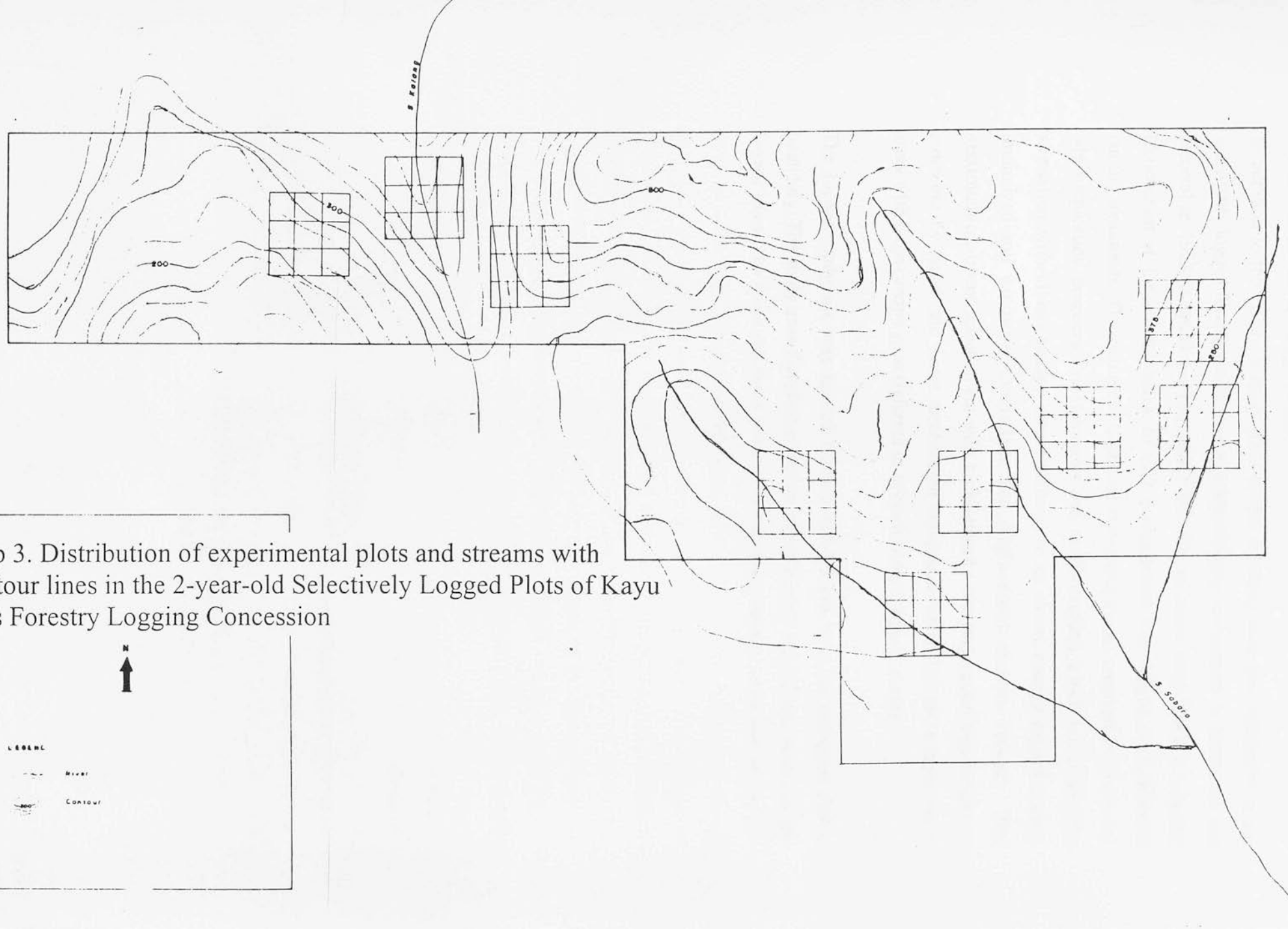
The UK-Indonesia TFM project is comparing 4 different selective logging regimes with a control (Proud 1995). Accordingly 15, nine hectare (300 x 300m) permanent sample plots (PSPs) were established (Map 2) and a detailed topographic survey of each was completed in 1993. The plots have been laid out with a central hectare reserved for intensive vegetation measurements. The central hectare is thus surrounded by a 100m buffer zone, and logging treatments applied to the entire surrounding 9 ha plot.



Map 2. Distribution of the experimental plots and streams with contour lines in the Permanent Sample Plots of Unlogged forests

The central hectare of each PSP has been subject to full botanical investigation of all tree species with DBH (diameter breast height) >10cm. Collection of specimens is complete and most have been identified to species by the Edinburgh Royal Botanical Garden. Pre-logging inventory has been completed for all stems with DBH>10cm. Repeat measurements yield estimates of incremental growth, mortality and recruitment in natural forest. The structure of the canopy has been recorded at ten locations in each plot using hemispherical photography. In addition, research describing the seedling and sapling pools is in progress giving baseline information for each plot. Automatic meteorological stations located above and below the forest canopy have provided microclimate data for more than two years. Associated research projects currently under way include topics such as rainfall interception, mycorrhizal associations and investigation of the impact of logging on the regeneration and growth of the forest ie. dipterocarp seedling physiology, response to changes in micro-environment and anurans respectively (Proud 1995).

The original timetable of this project was that logging would commence in 1994 and be completed by 1995. In accordance with this plan the experimental programme of this thesis initially assumed that pre-logging samples of amphibians would be taken from 9 plots (100m x100m for each plot) selected randomly in the PSPs, and that after logging was completed the same plots would be resampled. In accordance with this plan the pre-logging sampling took place in 1994/1995, but part way through this sampling it became apparent that the logging would not be completed on time. Indeed logging was not finally completed until 1997. For the purposes of this thesis which aimed to examine the impact of logging on amphibian populations and communities, it was necessary to identify an alternative area of selectively logged forest adjacent to the Permanent Sample Plots. After some considerable thought separate sites within the Kayu Mas Forestry Concession were selected as the logged 'treatment' (Map 3) to be compared with the sites at the Permanent Sample Plots which functioned as the 'unlogged control'. The logged site was selected in terms of its similarity in forest habitat, microclimate and topographic conditions with the PSPs.



Map 3. Distribution of experimental plots and streams with contour lines in the 2-year-old Selectively Logged Plots of Kayu Mas Forestry Logging Concession

2-year- old Selectively Logged Plots

In addition to the PSPs, 9 study plots (100m x 100m) were also delineated in the selectively logged forests within the Kayu Mas Forestry concession. In November and December 1993, a small portion (about 600 ha) of Kayu Mas Forestry Logging Concession at the cutting block of Z22 compartment was logged with different cutting treatments. The minimum-treatment selective cut area received low-intensity site preparation practices (by chainsaws and small tractors) which left much live vegetation with closed canopy cover (referred to as *closed canopy logged areas*), minimized soil exposure, compaction, and displacement of plant residue. The maximum-treatment selective cut with high-intensity site preparation was subject to more use by larger and faster mechanized equipment which left more open space areas with large canopy gaps (referred to as *open canopy logged areas*).

The logged study site was located between 0.5-1.5 km from the unlogged PSP's boundary. The study sites therefore comprise *closed canopy logged* and *open canopy logged plots*. The forest condition in terms of the closed canopy cover was almost the same as in the unlogged plots. In the latter, plots have less than 60% canopy cover and tree crowns form discontinuous canopy with large gaps. The dominant dipterocarp species in the lowland forest within the concession were bengkirai (*Shorea laevis*), meranti kuning (*S. hopeifolia*) and kapur (*Dryobalanops beccarrii*). The dipterocarp species in the logged site were very similar to the PSPs in terms of their forest structure and composition. Within the logging-disturbed lowland forest, five mutually exclusive structural types can be identified (*pers obs.* 1994). These were:

1. Open gap, areas in which there was no tree canopy within a 5 m radius, including low tangles of *Gleichinia sp.* (Gleichiniaceae) ferns and *Uncaria spp.* (Rubiaceae) lianas,
2. Short second growth, consisting entirely of fast growing, pioneer trees, none taller than 6m, mostly *Macaranga spp.*, *Glochidion spp.* (Euphorbiaceae), and *Trema orientalis* (Ulmaceae),
3. Tall second growth, same as (2), but with trees taller than 6m,

4. Sparse canopy fragment, a remnant portion of canopy in which most tree crowns were isolated or in clumps of only a few trees,

5. Dense canopy fragment, a remnant portion of canopy in which tree crowns formed a more or less continuous canopy, but which clearly lacked the full development evident in mature forest canopy.

Maximum daily temperatures recorded at the closed canopy logged site ranged from 27° to 31°C. In the open canopy logged areas the temperature was higher and more variable ranging from 34°C to 40°C and at night the temperature ranged from 17°C to 22°C. Minimum recorded relative humidity was higher in the logged areas of closed canopy (80%), and was less than 70% in the open areas of logged forest. Relief varied from hilly ground to steep and almost the same condition as the PSP's. More than 40% are hilly composed of steep slopes and rugged terrain in particular the western and southern parts. PHT Kayu Mas (1994) reported that soils of the logged sites had a surface horizon a few cm thick and there was an increasing risk of soil erosion with increasing disturbance and slope. The soil characteristic and its sensitivity prone to erosion in the logged site was similar with the PSPs condition (Proctor 1994).

3.6. Logging Systems and Their Impact

3.6.1. Introduction

Current selective logging operations in Indonesia invest heavily in haulage and transporting machinery. In such cases, a high extraction level is required to cover costs. This is afforded by the dense timber stocking in Indonesia: where extraction levels average 50 m^3/ha in Sumatra and can reach 90-120m^3/ha in Kalimantan compared with average levels of 72m^3/ha in Malaysia (Johns 1989) and 8.4 m^3/ha and 13.5 m^3/ha in Neotropical and African rain forests respectively (Baker 1984).

The overall extraction level in the study area (Z22 forest compartment) was 18.3 trunks/ha of marketable timber (24m3/ha basal area), a moderate level for hill dipterocarp forest in West Malaysia where extraction levels commonly reach 25

trunks/ha (Burgess 1971). In Southeast Asia up to 72 trees may be felled per ha (Whitmore, 1984), although extraction levels of 20-25 trees/ha are more representative (Johns, 1989). Elsewhere, for example in Gabon, Africa, White (1992) estimated that canopy loss (indicative of overall damage) averaged 10% with the extraction levels varying from 2 trunks/ha to 15/ha. In Ghana, Hawthorne (1991) recorded that an extraction level of only 1.6 trees/ha resulted in 20-30% canopy loss, leaving about 3% of the ground covered by roads and loading bays. The responses of the animal community to logging and their survival in regenerating forest will depend on the environmental and vegetational change induced by the logging process.

The system of logging used at the study site was a selective forest management system (for description see Whitmore 1984) adjusted to the Indonesian Selective Cutting and Replanting System (cutting trees with minimum girths 50cm). Logging is mechanized using heavy bulldozers to drag logs from slightly steep slopes to loading areas. The system relies largely upon natural regeneration of the forest, supplemented by replanting severely damaged areas with nursery grown trees. A fast-growing exotic species, *Eucalyptus sp.* or *Acacia sp.* as shade tree is currently used for this purpose in preference to local hardwoods in addition to the local species *Shorea leprosula* and *Gmelina sp.*

There is considerable debate concerning the extent to which deforestation may affect the environment on both local and global scales. It is beyond the scope of this thesis to examine global implications of deforestation, or even likely changes in climate within Indonesia. It is necessary, however, to consider local effects of changes in the vegetation cover, either by deforestation or by alteration of the cover as in the case of selective logging operations. Although the latter may seem considerably less drastic, there is little doubt that its consequences may be serious (Leighton and Irawan 1984).

3.6.2 Macro-environmental effects

Removal of forest cover over large land areas has been suggested to cause dramatic changes in local climatic conditions (eg. Potter *at al.* 1975, 1981, McElroy and Wofsy 1984), although little definitive data exist (Johns 1989). One well-documented effect

of moderate to heavy selective logging levels is the drying of the forest vegetation and an increase in the susceptibility to fire outbreaks (Uhl and Buschbacher 1985). The dangers of this were emphasized in the giant fires in Kalimantan following a period of drought induced by the aberrant 1982-83 El Nino event in the Pacific region (Leighton and Wirawan 1984). Logged forest was much more susceptible to fire outbreaks than adjacent unlogged forest. The implications of this for forest management of increasingly large areas of rain forest is obvious and it strengthens the argument for the preservation of unlogged forest patches within them.

3.6.3 Micro-environmental effects

3.6.3.1 Microclimate

Undisturbed rain forest is typically dark, humid, cool and wind-free, with scattered patches of early successional vegetation in treefall gaps and along water-courses. Recently-logged forest is well-lit, relatively dry, hot at midday, and experiences greatly increased wind turbulence. The proportion of early successional gaps is greatly increased in logged forest and large areas will be of similarly-aged trees, dating back to the time of logging.

Perhaps the most important constraint on the regeneration of natural forest is the degree of insolation. Many forest trees are shade-loving and will not germinate in open areas (Nicholson 1960, Whitmore 1984). In tall forest, 2% or less of the visible light striking the canopy reaches the forest floor and this consists largely of ephemeral sunflecks (Ewel and Conde 1976). In open clearings within logged forest, the level of insolation may be 84 times greater (Schultz 1960). This may have considerable implications for the re-growth of many tree species, including timber trees, many of which require different light regimes at different stages of their life cycle in order to achieve optimim growth (Becker *et al.* 1989).

The forest microclimate is influenced by canopy density and is characterized by a reduction in light levels, air temperature and increased humidity (Ashton 1992, Brown 1993). The physical environment within a forest is significantly altered by

logging activities. Logging systems create artificial canopy gaps with a variety of sizes, shapes and orientations that depend on the method and intensity of logging. The structure of the forest following selective logging is characterized by a complex mosaic of disturbance and forest types. These range from areas of relatively undisturbed forest under closed canopy, through areas with partial canopy on the margin of canopy gaps and finally large canopy gaps or open areas (Cannon *et al.* 1994). The creation of canopy gaps influences forest regeneration with a number of recent studies emphasising the importance of gap size for stimulating seedling growth (Brokaw 1985, Stewart *et al.* 1991, Chandrashekara and Ramakrishnan 1993).

The energy balance of the forest is altered because of increased penetration of radiation into the lower levels of canopy and to the forest floor. This in turn affects other aspects of the physical environment including air and soil temperature, humidity and the water balance or hydrology of the forest. Logging activities also result in soil disturbance and damage to trees remaining in the stand. The vertical temperature patterns in the forest are largely lost following logging (Chew 1968). The magnitude of these effects on the physical environment will depend on the system of logging and site characteristics such as slope, aspect and soil characteristics (Proctor 1994).

Differences in soil temperature are even more marked than in air temperature. Mean monthly soil temperatures in a forest clearing are both higher and more variable than in undisturbed forest (Schultz 1960). Temperature variation in the soil is largely dependent upon the degree of forest removal: that of a small treefall gap is less than that of a cleared area such as a large canopy gap area.

Early secondary vegetation does much to retard the evaporative power of air close to the ground, decreasing evaporation to one-third the level seen in large canopy gap areas, but this is still considerably higher than in the lower levels of unlogged forest. Loss of water from the litter and soil may be a critical factor affecting the soil microfauna and microflora: the water content of the soil decreased from 33% in a shaded plot to 28% in a half-shaded plot and 21% in an unshaded plot (Douglas *et al.* 1992).

3.6.3.2 Soil

The incidence of soil erosion, as a result of forest logging or clearance, has been much discussed (eg. Burgess 1971, Liew 1974, Soong *et al.* 1980, Hamilton 1985). Erosion can be caused as much by soil compaction as by soil exposure. In two studies in East Kalimantan, for example, the infiltration rate of water was found to decrease by up to 95% in areas affected by the passage of logging machinery (Hamzah 1978, Abdulhadi *et al.* 1981). Subsequent run-off can cause severe gulley and rill erosion on slopes.

Loss of soil can be severe. Douglas *et al.* (1992) found that a 35° slope in Sabah (North Kalimantan) lost 45 mm of the soil surface in six months, equivalent to 454 m^3/ha (but gained two-thirds of this amount from further up-slope). Therefore, even under selective logging, the silt load of forest streams can increase enormously and the sediment will be largely mineral rather than organic. Most soil loss occurs immediately following logging and re-establishment of vegetational cover cuts down soil loss considerably. Hamzah (1978) reports that three years after logging, the erosion level from a skidroad had been reduced from 12.9 to 3.2 tonnes/ha/month and water run-off from 189 to 19m^3/ha/month. The only exception to this trend might be steep slopes, where decay of tree roots over time causes a reduction in soil shear strength following logging and could increase the probability of landslips over time (Hamilton 1985).

3.6.3.3 Water

Total removal of forest cover generally results in a greater annual water yield through run-off and streamflow, since transpiration, atmospheric humidity and soil infiltration rates are all reduced. Partial selective logging of forested areas has been shown to yield 25% more run-off than was recorded in unlogged forests (Low and Goh 1972), but it has been stated that selective logging operations should not increase flood levels in major streams unless unusually large areas are logged simultaneously (Hewlett 1982).

The impact of logging on water quality largely reflects the techniques of logging and the care taken. During timber exploitation, streambeds may be used as skidroads, or crossings may be made of simple earth and log fills, which are left to erode away during subsequent rains, thereby increasing the silt level enormously (eg. Douglas 1968). Even under organized forest management operations, the silt content of rivers close to logging areas may be increased fourfold and that of streams within logging areas by tenfold (Hamzah 1978).

Water quality changes may result from increased mineral sediments or from increased organic matter content. The loss of organic material from small plots in Kalimantan was found to be 20 times greater in logged and up to 100 times greater in clear-felled areas (Kartawinata 1987). In addition, this may change water chemistry to some extent.

3.6.4. Effects of logging on vertebrates

Studies in Peninsular Malaysia and elsewhere have suggested that although a loss of 50% of trees caused by selective logging may have considerable initial effects on vertebrates, the level of resilience shown by the community was remarkably high (Wilson and Wilson 1975, John 1989, Crome 1991, Whitmore and Sayer 1992). Almost all vertebrate species present in unlogged forest had persisted or had recolonized by 12 years after logging (although the relative abundance of animal species was different in logged forests) (Whitmore and Sayer 1992).

In a few rain forests, logging levels are already in excess of 50% trees removed. In East Kalimantan-Indonesia, for example, the logging levels had reached more than 50% of trees removed in a highly selective logging during 1980 (Abdulhadi *et al.* 1981). It might be supposed that the abundance of many animal species would be correlated with damage levels. Large disturbances affecting rain forest will result mainly in the alteration of the relative abundance of species. There will, however, be a level of population reduction beyond which species cannot effectively recover and may become locally extinct, at least temporarily. The number of species whose populations are critically reduced will be a reflection of the severity of disturbance.

Mobile species, such as most vertebrates, may show the alternative response of simply moving away from a disturbed area and recolonizing later if conditions become suitable again (Karr 1986, John 1989). Small non-mobile species such as some amphibians are more likely to suffer severe population reduction as most cannot exist in heavily damaged areas around logging and log loading sites and neither can they migrate effectively (Blaustein and Wake 1990). The dispersion of such small species is likely to become patchy in logged-over forests. In time, this might well affect breeding success and even gene flow, unless the animals are able to re-occupy the regenerating forests and restore an even dispersion of individuals (Gascon 1991). It has been suggested that low humidity, increased temperature and lack of a permanent water source may be the critical factors permitting the survival of small vertebrates like amphibians (Corn and Bury 1989).

It is likely that the number of species whose populations are critically reduced by logging is proportional to the severity of disturbance. No supporting data are offered by this statement, but this may be due to an insufficiently large data set or to insufficient time having passed for changes to become fully expressed (less likely in the case of short-lived species). It may be predicted that species loss is a logarithmic function: at a high level of timber extraction and associated damage, the loss of a few more trees due to carelessness or bad management practices will cause disproportionately more losses than it would under low logging levels (Johns 1989). Following this argument, it may also be predicted that repeated logging, especially of small-sized trees, or re-logging before full regeneration has taken place, is likely to have severe and permanent effects on plant and animal diversity. Disturbance to complex ecosystems like rainforests is more often expressed in terms of changes in the relative abundances of species than in terms of species deletions.

In general terms, most reduction in abundance is seen among species specialized to exploit resources that suddenly become limiting, or species restricted by the microclimate changes observed in logged forest. Species occupying narrow niche widths have a higher probability of elimination by logging, and this is particularly marked among higher diversified groups such as insectivorous birds which are

speciated along both food resource and microclimatic gradients. In the case of primates there is no significant relationship between the population densities of common primates, or of their total biomass and the abundance of food (Yaeger 1984, Johns1989). A lack of correlation is also seen in the density of frugivorous birds and the abundance of food trees: hornbill biomass/km^2 in sout-east Sabahan forests is approximately equal over a range of forests from unlogged to heavily logged, despite a loss during logging of up to 73% of the original food trees (Leighton and Leighton 1983, Johns 1989).

Finally, it should be pointed out that these results relate to a fauna facing only a 50% tree loss as a result of logging. In Sabah-Malaysia and Sumatra-Indonesia, where vertebrates face a far more serious level of tree loss, with more severe problems of reduction in food availability and decrease in humidity, more species may be absent from recently-logged forest and recolonization may be slower. In such areas it is particularly important to assess the actual mechanism of recolonization: where do colonists actually originate and what is required to maintain a viable population of these species in the logged forests?

CHAPTER IV

RESEARCH METHODS

Three different sampling methods were applied in a study on the impact of selective logging on the abundance of anurans in the rain forests of Central Kalimantan. The methods were pitfall traps, searched quadrats, and night riparian transects. All these methods were only suitable for sampling terrestrial anuran species. They do not sample the arboreal species efficiently. The review of each method applied in temperate and tropical habitats is presented first and then followed by the sampling methods used in this study. This chapter also presents the summary of sampling activities in the study sites and describes the vegetation sampling and microclimate measurement. Statistical analyses used in evaluating the data are also given.

4.1. Introduction

4.1.1. Drift fences

Straight-line drift fences typically are short barriers (5-10m) that direct animals travelling on the substrate surface into traps placed at the ends of or beside the barriers. Drift fences with pitfall traps are commonly used to inventory and monitor populations of amphibians and reptiles (Corn and Bury 1990), yielding data on species richness, relative abundances and habitat use of selected species (Bury and Corn 1987).

Both arrays of drift fences and grids of individual pitfall traps have been used to sample terrestrial amphibian assemblages in a variety of temperate habitats, including deciduous forests (Pais *et al.* 1988), coniferous forests (Jones 1988), riparian woodlands (Friend 1984) and wetlands (Beauregard and Leclair 1988). However, Inger (*pers.comm*) stated that pitfall traps and drift fences were not suitable to capture Bornean anurans in Dipterocarp forest at Bukit Lanjan, West Malaysia. They were not sampling the species equally and only a small number of anurans were captured by these pitfall traps. Inger has commented both on the

extreme scarcity of forest litter animals of all kinds in Bornean forests and the lower number of anuran individuals. For example, Inger and a team of collectors were only able to amass an average of 20.5 amphibians per day, whilst in Costa Rica and Panama, a single collector can catch as many as 50-100 amphibians in a day (Scott 1976).

Drift fences and pitfall traps do not sample all species equally (Karns 1986, Corn and Bury 1990, Dodd 1991). Surface-dwelling species that have limited climbing, jumping, or burrowing abilities, such as mole salamanders (*Ambystoma* spp.) and some anurans (*Pseudacris ornata*, *Bufo* spp.), are best sampled by the drift fence technique. Sampling of anuran species with strong jumping and good climbing abilities (e.g., *Acris, Gastrophyne*, most *Rana* and *Hyla*) is far less efficient (Franz and Ashton 1989, Dodd 1991). However, in some cases pitfall traps can efficiently capture a certain species with good climbing ability. For example, several studies report the capture of large numbers of eastern narrow-mouthed toad (*Gastrophyne carolinensis*) using drift fences (Campbell and Christman 1982, Enge and Marion 1986, Mengak and Guynn 1987). Because of the climbing ability of this toad, however, the numbers of individuals captured probably do not accurately reflect its relative abundance.

The length of the drift fence influences the number of animals captured, and the optimum length varies with habitat type. Bury and Corn (1987) compared arrays of 2.5-m fences with those of 5-m fences and found that those of 5-m fences captured more amphibians than 2.5-m fences in coniferous forests in northwestern North America. So far, too little experimentation on the efficacy of different array designs has been done and too much variation exists among amphibian assemblages to select and recommend a single design suitable for all situations (Heyer 1994). However, a four-fence array radiating outwards at 90° angles with pitfall traps at the end of traplines as suggested by (Dalrymple 1988) is probably suitable for most studies.

4.1.2. Quadrat Sampling

Quadrat sampling consists of laying out a series of small squares (quadrats) at randomly selected sites within a habitat and thoroughly searching those squares for amphibians. This technique can be used to determine the species present in an area, their relative abundances, and their densities (Inger 1980).

Quadrat sampling has been used most effectively for sampling amphibians in the forest litter, where species often occur in high densities but are difficult to detect because of their secretive habits. The method has reduced effectiveness in habitats with dense ground cover or steep terrain where it is difficult to place quadrats on the ground at random (Heyer *et al.* 1994). Although the method has been used in relatively open vegetation, it is most effective in closed-canopy forests. Quadrat sampling has been effectively employed in tropical forests to determine density, species diversity, and relative abundances (Lloyd *et al.* 1968, Scott 1976, Inger 1980 and Lieberman 1986), and in temperate forests to estimate densities of salamanders over long time periods (Jaeger 1980, Mathis 1990).

Two methods for quadrat sampling have been used for sampling amphibians. Point sampling (small quadrats of 1x1 m) has been used for studying single species with small, densely distributed populations (e.g., the salamander *Plethodon cinereus*, in which density can reach 3 individuals/m^2). Broad sampling (large quadrats measuring either 5x5 m or 8x8 m) has been used both to sample species in which individuals are widely spread and when populations consist of multiple species (Heyer *et al.* 1994).

4.1.3. Night Riparian Transect

Amphibians frequently respond differentially to environmental gradients, especially gradients that reflect moisture (Corn and Bury 1990). Transect methodology can be used to sample either across these habitat gradients or within habitat types. Transect sampling using a randomized design can help track species numbers, relative abundances, and densities across habitat gradients. The method is very useful in determining intraspecific and interspecific changes in amphibian populations across

some continuously changing environmental feature (Heyer *et al.* 1994). Night riparian transect sampling is one of the best techniques for studying habitat gradients in the streams of lowland areas (Inger 1969). Night riparian transects have also been used for precise mapping of distributional and habitat discontinuities between species (Jaeger 1970), and for tracking changes in numbers and densities of salamander species along gradients from streams to upland areas (Hairston 1949, 1980).

4.2. Sampling methods used in this study

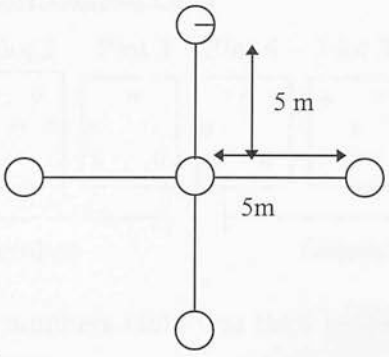
I used three different methods to sample anurans: riparian transects surveyed along streams, randomly placed quadrats and pitfall traps on the forest floor. All methods were used in both the unlogged forest of the Permanent Sample Plots and in the 2-year-logged forest of Kayu Mas Forestry Concession. Sampling with riparian transects at night and searched quadrats by day lasted two to three hours unless interrupted by heavy rains. Night searches consisted mainly of localizing individuals after hearing their calls or through direct observation, in particular for the species that are riparian. Quadrat investigation consisted of examining refuges, for example decaying trees, leaf axils, fallen leaves or logs, tree holes and forest litter for the amphibians in forest floor plots. The active searched quadrat was concentrated mainly on species of anurans that use streams only for breeding and species that wander widely through the forest at all ages (not tadpoles) (see page 34 and 35 for the Ecology of Bornean anurans).

4.2.1. Research Design for Pitfall Traps

The entire trap comprises 4 trap lines radiating outwards at 90 degree angles from a central trap. This design was a modification of that outlined by Dalrymple (1988). Each trap line consisted of a 5m-long drift fence, the maximum uninterrupted length feasible in dense forest. Longer fences were not feasible because of many obstacles (e.g. logs and trees) in the forest. The drift fences were made from plastic sheeting 30cm high, delimiting 4 areas within which anurans were guided in a direction leading to pitfall traps either at the center or the perimeter of the trap. This array of 4

5-m arms, providing a total length of 20m per array, covered an area of 100m² (Figure 4.1)

Figure 4.1. Design for arrays of drift fences with pitfall traps

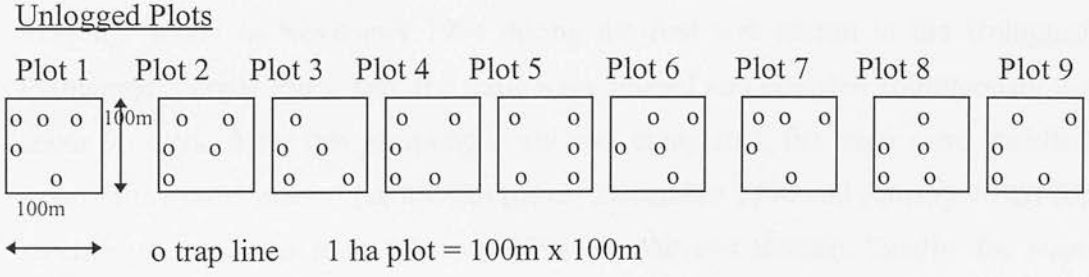


5-m drift fence ——

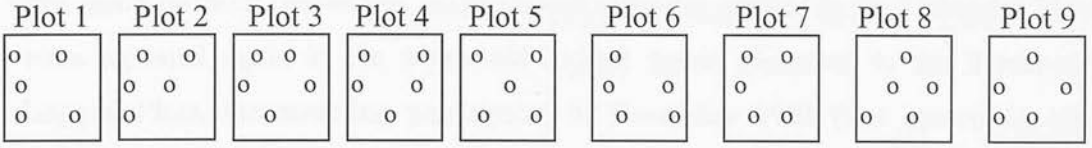
pitfall trap ◯

Each pitfall trap consisted of a plastic bag. Each bag had a perimeter mouth of 20cm and was 60cm deep, and the traps were sunk into the ground to 60cm depth. Five trap lines were installed randomly in a 1-ha plot. There were three replication plots every month and each plot consisted of five trap lines (Fig 4.2). So, in total there were 15 trap lines per month observation. Pitfall traps were placed at the ends of the fence so that no gaps occurred between the fence and the rim of the trap. The traps were protected from the rain by 30x30cm plastic sheeting and were placed at an angle at the end of the drift fences. These covers were only marginally successful at keeping the traps dry. No water was put in the traps, because this has a deleterious effect on the preservation of amphibians (Bury and Corn 1987). The lay out of trap lines is shown in Figure 4.2.

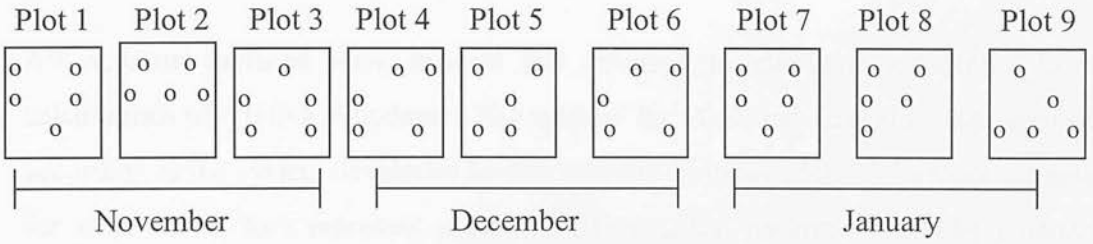
Figure 4.2. Layout of trap lines



Closed Canopy Logged Plots



Open Canopy Logged Plots



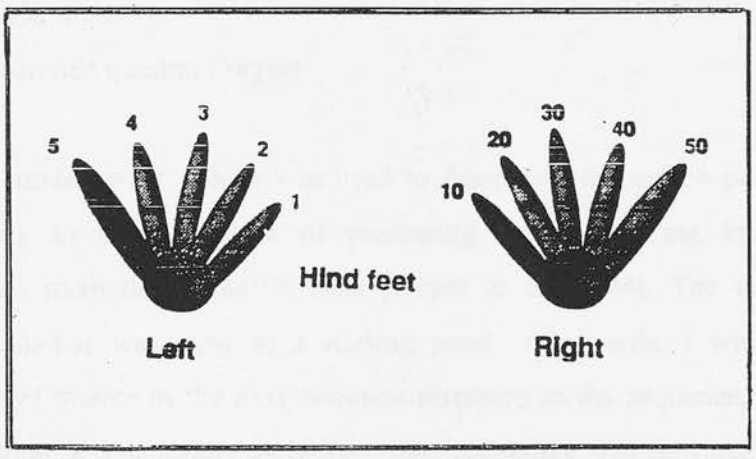
A random numbers table was used to determine the positions of trap line arrays in each plot. The starting point was usually from the SW end (corner) of the plot. Then, the first two-digit number obtained from the table of random numbers read as the x (horizontal) axis in m and the next two-digit number read as the y (vertical) axis in m, the joint point between the x and y axis was the centre of the trap line. Thereafter, a randomized sequence of sampling was used to determine the position of the next trap line array starting from the SW again. For example, random numbers were chosen with 4 digits such as 1576. I then walked (started from the SW end of the plot) to the horizontal distant as far as 15 m, afterwards I walked as far as 76 m to the vertical direction, the joint point was the centre of the trap line.

Traps were checked daily, early in the morning or before noon on a regular basis. As the trapping rate declined over time, the interval between checks increased to about three days. The following information was recorded at the time of capture for each specimen: date, time, humidity and rainfall, microhabitat and circumstances of capture. The species (also the individuals, size and maturity), the array and trap numbers of all individuals caught were also recorded. The first three months of trapping started in November 1994 during the first wet season in the Unlogged Permanent Sample Plots, and, the traps were opened and operated continuously for about 90 days. After this sampling event was completed, the traps were installed again in the same places (as for November, December 1994 and January 1995) for another three months from February 1995 (second wet season). Finally, the traps

were installed again in early May for three months during the dry period in 1995. The same research procedures, methods and activities applied in the Unlogged Plots were repeated again in the 2-year-old logged forest. However in the 2-year-old Logged Plots, the sampling was started in November 1995 (wet season 1), and continued in February (wet season 2) and May (dry season) 1996.

All anurans captured were marked and released to eliminate recaptures from calculations of relative abundance. The method for Mark and Recapture was applied according to the system developed by Martof (1953). In the Martof clip-code scheme for amphibians, toes represent numbers. The smallest number is usually counted from the inner part of the hind feet toes. For example, to mark anuran number 13, one toe (the first) of the right hind foot and one toe (the third) of the left hind foot must be cut (see Figure 4.3, clip-code scheme for marking anurans).

Figure 4.3. Numbering system for marking the Hind feet of an anuran



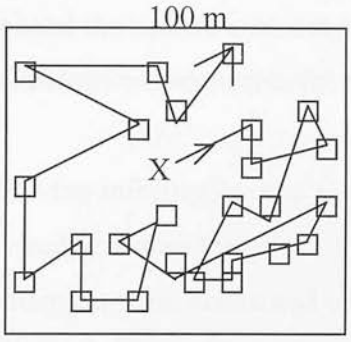
4.2.2 Research Design for Forest Floor Plots

Searched quadrats using the Broad Sampling Technique (5x5m) was applied according to the standard methods for amphibians (Heyer *et al.* 1994). In this technique, the first sample point was determined by using a random number table (the same procedure used for determining the first starting point for locating the position of trap line arrays in each plot), then a sequential series of random sample points was chosen on the forest floor. Twenty five sample points were distributed randomly in a 1-ha (100mx100m square plot). A 5 x 5m quadrat was laid out using

stakes and twine, thus covering an area of 25-m squares. An example of the experimental design used for searched quadrats is shown in Figure 4.4.

Figure 4.4. General sample of search pattern

Figure 4.



X The first sample point

Walk direction

searched quadrat (5x5m)

A randomized walk design was used to determine the search pattern in each plot according to the technique of measuring and monitoring biological diversity (standard methods for amphibians (Heyer *et al.* 1994). The centre of the first square/quadrat was used as a starting point. Afterwards, I walked for the next number of metres in the next compass direction in the sequence, again established the quadrat and sampling as before. This procedure was repeated for a total of 25 sample points.

After selecting the quadrat locations and before the searching began, one person on each side of the quadrat removed all litter from a 50cm swath along the outer perimeter in order to make an escaping animal easily detectable and to minimize possible bias due to edge effects. This clearing technique followed that outlined by Lloyd *et al.* (1968) for the Kalimantan anuran studies, except that the soil surface was not turned over by a hoe. The first series of observation plots were actually done

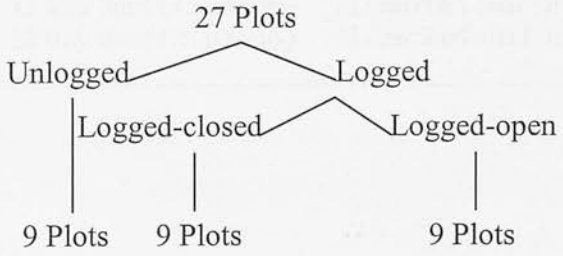
by hoe, but less than one animal per plot was located in this way and the practice was abandoned. After clearing the edges of the quadrat, all litter and ground cover inside the quadrat was removed and the number of individuals of each amphibian species present were counted. This work was undertaken by first examining strips inside the quadrat and parallel to the boundary twine. Successive strips were worked from the outside toward the centre, until the entire area was covered. This procedure was repeated, in a randomized sequence, for all quadrats.

In addition, the following information was also recorded for each quadrat sampled:

- location of the quadrat within the grid;
- date, time at which sampling began and was completed;
- temperature and relative humidity;
- vegetation type in the quadrat;
- slope of area on which the quadrat is located (using a clinometer);
- estimated canopy cover (as a percentage of area directly above the quadrat);
- estimated shrub cover (percentage of the quadrat covered by multistem plants > 1m);
- estimated herb cover (percentage of the quadrat covered by herbs and seedlings < 1m tall)
- estimated leaf litter cover (percentage of the quadrat covered by leaves)

The same methods, work procedures and time of observations were also applied in the logged forest as in the unlogged forest (Permanent Sample Plots).
The sampling regime for pitfall traps and searched quadrats is summarized in Figure 4.5.

Figure 4.5. Sampling Regime for forest floor sampling



Each site consisted of 9 observation plots. The logged site was split into two parts, Logged-closed plots and Logged-open plots. So, in total there were 27 observation plots sampled by pitfall traps, drift fences and searched quadrats. The first three months of quadrat sampling started in November 1994 during the first wet season (75 searched quadrats per month observation) in the Unlogged Plots, and, the quadrat sampling work was conducted for about 90 days (November-January). After this sampling event was completed, the next quadrat sampling was begun in February 1995 for the second wet season (February- April). Finally, quadrat sampling was carried out again in early May for three months during the dry period (May-July) in 1995. The same work procedure and sampling technique were repeated in the 2-year-old Logged Plots but the dates of sampling were from November 1995 till July 1996. A summary of overall sampling activity is presented in table 4.1.

Table 4.1. Summary of activities in the Unlogged Plots (Permanent Sample Plots) and the 2-year Logged Plots at Camp 92, Central Kalimantan. Sampling effort in the logged plots was distributed in the same way, but started in November 1995 and finished in July 1996.

Plot(Time)/Activities	Pitfalls/ Drift Fences	Quadrats	Stream Transects
Unlogged Plots (PSPs)			
Wet SeasonI:			
November 1994	15 trap lines (75 pitfalls)	75 searched quadrats	8 night surveys
December 1994	15 trap lines (75 pitfalls)	75 searched quadrats	8 night surveys
January 1995	15 trap lines (75 pitfalls)	75 searched quadrats	8 night surveys
Wet SeasonII:			
February 1995	15 trap lines (75 pitfalls)	75 searched quadrats	8 night surveys
March 1995	15 trap lines (75 pitfalls)	75 searched quadrats	8 night surveys
April 1995	15 trap lines (75 pitfalls)	75 searched quadrats	8 night surveys
Dry Season:			
May 1995	15 trap lines (75 pitfalls)	75 searched quadrats	8 night surveys
June 1995	15 trap lines (75 pitfalls)	75 searched quadrats	8 night surveys
July 1995	15 trap lines (75 pitfalls)	75 searched quadrats	8 night surveys

4.2.3. Research design for stream transects

Night riparian transects were carried out on six streams flowing through the Unlogged Plots of the Permanent Sample Plots (Map 1) and in the 2-year-old Logged Plots (Map 2). These streams were part of the typical tributaries of the Kalang river and they are located at Camp 92, Central Kalimantan. The average width of the sampled small streams was about 4m and the large streams was 8m. The small stream widths varied between 3-5 m and the widths of large streams varied between 7-10 m. I measured stream width and depth of riparian habitat surveyed at 3 intervals (0, 5 and 10 m) by a diameter tape. Depth at each interval was calculated as the sum of the depths at three points (1/4, 1/2, and 3/4 of the width) divided by 4 (Platts *et al.* 1983). Mean depth and width were used in the analysis. A clinometer was used to determine the side slope (%) adjacent to each stream. The particle size-class of the substratum that was most abundant in a transect across the stream at each 10-m flag (15 observations per stream) was estimated by eye. Substratum particle size classes were silt, sand (particle size 1-2mm), gravel (3-32mm), pebble (33-64mm), cobble (65-256mm), and boulder (>256mm) (Corn and Bury 1989).

At each plot site on stream banks, riparian night stations were marked with plastic flagging. The distance between successive stations was 10m. From all points, at least two marked positions were visible. The observational areas were mapped by using a compass. There were 15 reference stations distributed along the stream from down stream to up stream. The transects were placed in equal numbers and sampled in random order . In total a surveyed strip of 150m long was laid out. The night riparian survey was carried out and repeated four times in each small stream and large stream as well. So, there were eight night riparian transects per month observation.

A study team equipped with flashlights walked up a stream bed searching both banks. We searched exposed portions of the bed, and the vegetation along the banks. Every anuran seen was recorded, captured and its position noted (either horizontal or vertical) and the distance of each observed anuran relative to the stations was also

noted. The substrate condition and vegetation type were also noted. The Martof clip-code scheme (1953) was used to mark anurans and estimate their population size. Anurans captured within the survey strips were marked and released but anurans smaller than 40 mm (except for microhylids) were rarely marked because of the difficulties in clipping toes of small anurans. If the anuran had been marked previously, its number was recorded.

When captured an anuran was identified to species, measured, marked (unless already marked), its position recorded and released at the point of capture. The capture point was determined by compass with reference to the nearest survey station. This distance was measured using a measuring tape. Distance from the edge of the stream bed was also recorded. Measurements of the size from the tip of the snout to the end of the body (referred to here as snout-vent length) were made by a digital dial caliper marked in mm.

4.3. Vegetation Sampling and Microclimates Measurement

In unlogged and logged forests, vegetation components within the pitfall array were sampled and divided into 4 square quadrats (5x5m per each quadrat), with the center of each sampling plot equidistant from the center trap. Vegetation variables (ie. canopy cover, shrub cover, herb cover, and litter cover) in the plots were recorded and the percentages of the total surface area they covered were visually estimated with estimation errors approximately ± 5%. In addition, tree species, girths (measured by a diameter tape) and number of trees within 10x10 m square plots were recorded. The height and girth of all trees larger than 10cm DBH in the plots were measured during wet season 1, 2 and the dry season. Most of the trees in the PSPs have been identified by botanists from Edinburgh Royal Botanical Garden but unrecognised species found in the logged plots were identified by using a tree field guide manual for Indo-Malay flora (Richard, 1985). The same work procedure and vegetation sampling were also carried out in the 5 x 5 m searched quadrat plots.

Climatic variables such as rainfall, humidity and temperature were recorded in each site. Micrometeorological data were obtained from automatic weather stations (CR 10, Campbell Scientific (UK) Ltd., Leicester, UK) located in a closed canopy gap at a representative plot of the Permanent Sample Plots and in a large canopy gap at a representative plot of the Logged Plots. The data from these weather stations (Campbell CR 10 loggers) were connected to a card storage module/data logger (Delta-T Device Ltd., Cambridge, UK). The data were then automatically transferred to a removable card, which was carried to the project camp and read using an identical storage module connected to a portable computer (TP 720, IBM Corp., USA) once every two weeks. In addition, three rain gauges, thermometers and humidity meters were placed in each study plot. Rainfall data for the same period of measurement were read manually from the plastic container rain gauges using a 100 mm measuring cylinder. The rain gauges were placed under the canopy gap areas of the unlogged and logged plots. The temperature and humidity at a height 1 m above the ground were recorded at the same time when we checked the pitfalls and observed the searched quadrats. The thermometer and humidity meter (protected from rain by a man made roof) were set up under the canopy gaps at 1 m height off the ground at each plot. Soil temperature in the study plots was also recorded by using soil temperature meters. The soil temperature was measured directly in the searched quadrat at approximately 10 cm deep at the edge and centre of the quadrat, and also where any anuran was found. Water temperature was measured at approximately 15 cm either in the vicinity of any amphibians which were located or approximately 0.5 m out from the edge according to a Standardized protocol for surveying aquatic amphibians (Fellers and Freel 1995).

4.4. Statistical Analysis

Raw data for the micrometeorological variables were processed using various computer software packages. Spreadsheet software (Microsoft EXCEL, Microsoft Inc., USA) and a statistical package (MINITAB, Minitab Inc., State College, PA, USA) was used to analyse the field data, while graphic software (Microsoft EXCEL)

was used to produce different types of graphs. Data comparing the abundance of anurans and their species richness between sites in different seasons was analysed using the General Linear Model (GLM) in MINITAB. In addition, regression analysis was undertaken to investigate the relationship between dependent (ie. anurans abundance and species richness) and independent variables (ie. rainfall, humidity, temperature and vegetation). Association between abundance and species richness and categorical vegetation and microclimatic variables was analysed through analysis of covariance (ANCOVA). Measurements of percentage cover were arcsin-transformed (Sokal and Rohlf 1995) before analysis.

Sorenson's (1948) and Simple Matching (Sokal and Michener 1958) coefficients for species richness and abundance were used to assess the degree of similarity of species composition within a selected habitat or sample. These data can provide information on species interactions that may be particularly useful in explaining observed distributions. Such interactions were then examined by means of pairwise comparisons among species in a given area. Fourfold (or 2 x 2) contingency tables were used for summarizing data regarding the relationship between two species, populations or habitats. Multivariate Analysis with Principle Component Analysis (PCA) was applied to describe the two-dimensional representation between variables and to analyse species composition in the area concerned.

CHAPTER V

IMPACTS OF LOGGING ON VEGETATION AND MICROCLIMATE

5.1. Background Information

5.1.1. Effects of road building damage on vegetation

The heavy hauling machinery used in logging operations within Kayu Mas Logging Concession required a road system, the development of which had highly damaging consequences on forest structure and future regeneration. Up to 0.5 km/100 ha of primary surfaced roads were built to provide truck access. All cut boles were dragged to these roads. Roads were 5 m wide, and at least 20 m on either side is deforested too. Quarries were also opened to scrape the laterite for the road surface, and large parking areas were cleared to stack logs and load the trucks. Inundated zones along road embankments built without enough culverts to provide adequate drainage caused all trees to die. Branching out from these primary roads were secondary unsurfaced tracks, widened and deeply ploughed by broad cuts from repeated passage of buldozers and skidders. On these 5-6m wide tracks, all vegetation was removed and the heavily compacted soil exposed to rapid erosion. The road on steeper slopes near the streams in the logged plots is a more recent construction but was already eroding badly in 1994. In one place the road across a bridge had lost a third of its width and was rapidly approaching a dangerous condition. Gully erosion by the road side abounds and there are several ponds caused by the blocking of drains with eroded material. In other places, the logging road is at a gradient of 30% which is excessive and it is losing its surface layers. It is a small indication that the bedrock is already being exposed. Understory vegetation (trees up to 20 m high) is flattened out on these smaller 3m wide paths, and many trunks alongside are damaged by the vehicles. Finally, the gaps created by the felled trees are often enlarged by the skidder maneuvering to seize and pull out a log. Moreover, the trees felled to cut the tracks are pushed away from the bulldozer, so

that an area of forest wider than the opening is usually damaged by tree falls on either side of roads (pers.obs.).

Considerable information on vegetation in the logged plots has been collected by Kayu Mas Logging Concessionaire. Within the 25 1-ha sample quadrats of 2-year-selective logged forest in the Kayu Mas forestry logging concession, the vegetation at ground level was heavily damaged on an average of 38% of the area, divided into 30% for tree fall gaps and associated disturbances and 8% for tracks between them. The damaged areas were either devoid of any standing trees or were covered with a thick layer of slash (PHT Kayu Mas 1994). Similar levels of damage (33 - 54% of the total area) were measured by the Central Kalimantan Forest Service Agency on nearby forest logging sites under two different logging intensities (Schmitt 1990). Overall, 30% of the vegetation at ground level in the west Kalimantan lowland dipterocarp forest was disrupted by roads, tractor tracks and skid trails. Forest disturbance at ground level by logging equipment was strongly associated in space with heavy disturbance to forest structure; 80% of the total length of roads, trails and tracks were found in the 45% of this lowland forest area classified as heavily disturbed (Cannon *et al.* 1994). In West Malaysia, forest disturbance at ground level resulted in 8.4 % loss due to road building and log loading and 39.2% during felling and skidding (Johns 1988). In Southeast Asia, Ewel and Conde (1976) stated that major, skidder and extractor roads can cover up to a third of a site where heavy machinery is used during logging.

The mean gap size was markedly larger in logged than in unlogged forest because artificial gaps all came from the largest, fully crowned trees and were often enlarged by dragging manoeuvres, whereas many natural gaps were created by the tree falls of dead trunks. Skorupa (1986) recorded 20 and 40% decreases in canopy cover in lightly (extraction level of 2-5 trees/ha) and heavily (extraction level of 25 trees/ha) logged forest respectively in Kibale, Uganda, compared to a control, recorded about 15 years after logging. In the closed canopy logged plots of this study, canopy cover below 10 m was around 50%, slightly higher than in the site before logging. Levels at 10-20 m and > 20 m were slightly decreased (75% and

65% respectively), although overall cover was almost similar (95%). In this study, the harvest removed 65% of dipterocarp basal area in the open canopy logged plots, and resulted in heavy disturbance to 45% of the forest canopy. This 45% was in open gaps and dominated by early pioneer species at the time of sampling. These results indicate a somewhat higher disturbance level than that recorded by Abdulhadi *et al.* (1981), who reported 30% of logged-over forest left as open gaps.

5.1.2. Effects of logging on forest structure

Selective felling of timber trees leaves an irregular pattern of fragments with areas of almost undamaged forest, open gaps where trees have fallen and partially open skidroads. Although only the largest trees of particular species may be felled for their timber, the high level of associated damage to the remaining small timber trees and non-timber trees of all sizes means that the relative frequency distribution of tree height classes does not change following even quite intensive logging, at least for trees of 30 cm girth (Johns 1988). Similar results were obtained in the transect sample of the logged plots as indicated in Table 5.1.

Table 5.1. Distribution of trees within different height classes before and after logging in cutting block of Z22 compartment, Camp 92, Central Kalimantan (PHT Kayu Mas 1994)

Height Class (%) (m)	Before Logging (%) (N=1122)	After Logging (N=543)
< 14.5	38.6	35.5
15-19.5	32.7	33.7
20-24.5	16.5	19.1
25-29.5	5.6	5.7
30-34.5	4.4	4.7
35-39.5	1.5	0.9
>40	0.7	0.4

An analysis of the reduction in basal area of tree taxa resulting from intensive logging suggests that the proportional representation of different tree genera in the total basal area of the stand does not change either (Johns 1988). Basal areas of logged forest are, of course, very much less than unlogged sites. Since loss of trees

is random between size classes, reduction of basal area is in proportion to damage levels. However, logged forests often accrue to the 35 m^2/ha or so characteristic of unlogged forest fairly rapidly. For example, the mean height and trunk diameter of the commonest trees were 2 m and 2 cm respectively in one year-old logged dipterocarp forest, but 22 m and 35 cm in 10 year-old logged dipterocarp forest (Hue 1981). In Indonesia, many colonizing trees achieve a girth of > 30 cm within 5-6 years and a lower canopy may be re-established after 12 years (Cannon *et al.* 1994).

Loss of marketable trees is, however, only a minor effect of selective logging. Incidental damaged caused by construction of roads and log-loading areas, and by cut timber trees pulling down their neighbours, results in far greater losses. The extent of tree mortality varies among local habitat types, and is usually greatest on ridgetops (where major access roads, spar trees, and log-loading areas are located) and less on steep slopes and in the valley bottoms. The causes of tree mortality during the logging process can be shown in terms of loss from sample transects in Z22 compartment made by the Kayu Mas Logging Team (Table 5.2).

Table 5.2. Estimated causes of tree mortality in the cutting block of Z22 compartment in Lowland Dipterocarp forest, Camp 92, Central Kalimantan (PHT Kayu Mas 1994)

Damage/Loss Category	% total number of trees
Killed:	
Timber trees	4.3
Destroyed during road building	5.5
Destroyed during construction of spar-tree sites and log-loading areas	3.9
Destroyed during felling operations and log dragging (Incidental damage)	40.4
Total % trees killed	54.1
Standing but damaged	3.6
Total trees killed + damaged	57.7

Notes: Only boles of marketable trees were removed from the site. Trees killed during road building and clearing operations were bulldozed aside. Some trees killed during felling were uprooted, others suffered snapped boles and a few were killed by excessive bark damage.

Loss of 54.1% of the stand seriously disrupts the forest structure, creating many large gaps (open canopy) and breaks in the canopy. Most of the trees were killed (>40%) incidentally during felling operations and log dragging. Typical figures for incidental loss in other parts of the World are around 50% at equivalent logging intensities (e.g., Ewel & Conde 1976, Johns 1983, Whitmore 1984). Marn (1982, cited in Johns 1989) showed that directional felling and careful siting of skid roads could reduce levels of incidental damage by 50% in Sarawak, and Buenoflor & Tiki (1989) reported similar findings in Papua New Guinea.

5.1.3. Effects of logging on forest composition

Forest tree species diversity is consistently less in logged forest, the degree of reduction being roughly proportional to the intensity of logging (Johns 1989). Since a very small proportion of trees actually destroyed by logging were extracted for their timber, the term 'selective logging' is misleading. In Peninsular Malaysia dipterocarp forest, the logging process did not differentially affect the relative proportions of different tree families in the forest (Borhan *et al.* 1989). Similar results were measured in the transect sample of the logging plots (PHT Kayu Mas 1994) as shown in Table 5.3.

Table 5.3. Predominant tree families in the transect sample before and after logging (PHT Kayu Mas, 1994) at Camp 92, Central Kalimantan

	Before Logging		After Logging	
Rank	Family	% Sample	Family	% Sample
1	Euphorbiaceae	26.2	Euphorbiaceae	27.0
2	Dipterocarpaceae	8.5	Dipterocarpaceae	7.9
3.	Burseraceae	7.8	Burseraceae	6.9
4.	Myristicaceae	6.5	Myristicaceae	6.3
5.	Leguminosae	5.4	Leguminosae	5.3
6.	Meliaceae	5.1	Meliaceae	5.0
7.	Annonaceae	4.7	Annonaceae	4.4
8.	Sapindaceae	4.6	Sapindaceae	4.6
9.	Lauraceae	4.1	Myrtaceae	3.6
10.	Anacardiaceae	3.1	Anacardiaceae	3.3
% trees in top 10 families		76.0		74.3

Only the family Lauraceae disappeared from the top 10 families (table 5.3) after logging, and this was replaced by family Myrtaceae. A high correlation existed between the abundance of different families (Anonymous, 1994) in the sample before and after logging (Spearman rank correlation: $r_s = 0.97$, $N = 39$, $P < 0.01$), indicating that selective logging, at least at this intensity, resulted in essentially random destruction. Given that the effects of intensive logging are essentially random, rare trees will be most susceptible to deletion. Of course, a rare species that is also a highly valued timber will almost certainly be removed completely. This may be happening with such species as Jelutung (*Dyera costulata*), family Apocynaceae (Johns 1989).

5.2. Results

5.2.1. Effects of logging on vegetation cover

There was no difference in percentage of litter cover between the unlogged (88.6%) and the closed canopy logged plots (86%) in the study sites, but the open canopy logged plots had a much lower percentage of litter cover (57%) compared to the two previous sites. Herb cover (herbs and seedlings < 1 m tall) was higher in the unlogged (86.5%) and closed canopy logged plots (84.4%) but less in the open canopy logged plots (73%). Dominant herb cover in the unlogged plots was similar to the closed canopy logged plots consisted of bluestem (*Andropogon spp.*, 40%), wiregrass (*Paspalum spp.*, 10%) and low panicum (*Dichothellium spp.*, 5%). Dominant herb cover in the open canopy logged plots was wire grass (50%), low panicum (20%), and bluestem (5%). Percent cover by multistem plants > 1 m tall (shrubs and young trees) only varied slightly between unlogged (55%) and closed canopy logged plots (60%), but they had less cover than the open canopy logged plots (65.1%). The young individuals of *Macaranga sp.*, *Polyalthia sp.* and *Melastoma sp.* were dominant. The average canopy cover in the unlogged (82.3%) and closed canopy of logged plots (77.2%) was greater than the open canopy logged plots (65.3%). There was no significant difference on vegetation variables between unlogged and closed canopy logged plots (GLM ANOVA, $P > 0.05$), but the difference was highly significant between open canopy logged plots in comparison

to unlogged plots and closed canopy logged plots, respectively (GLM ANOVA, $P < 0.01$).

5.2.2. Precipitation, humidity and temperature

The climate of the rain forest in the study site is typified by having remarkable dry season with an average temperature of c. 25°C and a fairly constant relative humidity between 85-95%. The amount of rainfall varies considerably ranging annually from 2,800 to 3,900 mm/year. Figures 5.1 and 5.2 show that there was a remarkable dry season (ie. July and August) in 1995 in the Unlogged and Logged Plots. The difference of rain fall between wet and dry seasons was quite contrast. For example, monthly precipitation during wet season ranging from 250 to 500 mm and less than 100 mm during dry period in the unlogged plots. The average humidity was also different between wet and dry seasons. The average humidity around 91% during dry season and between 93-96 % during wet season (Figure 5.1).

Fig 5.1. Relationship between rainfall and humidity (average) during wet and dry period (1994/1995) in Unlogged Plots at Camp 92, Central Kalimantan

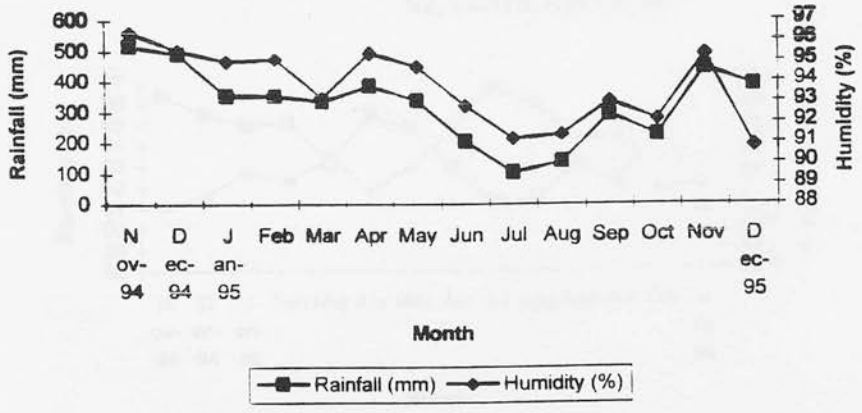
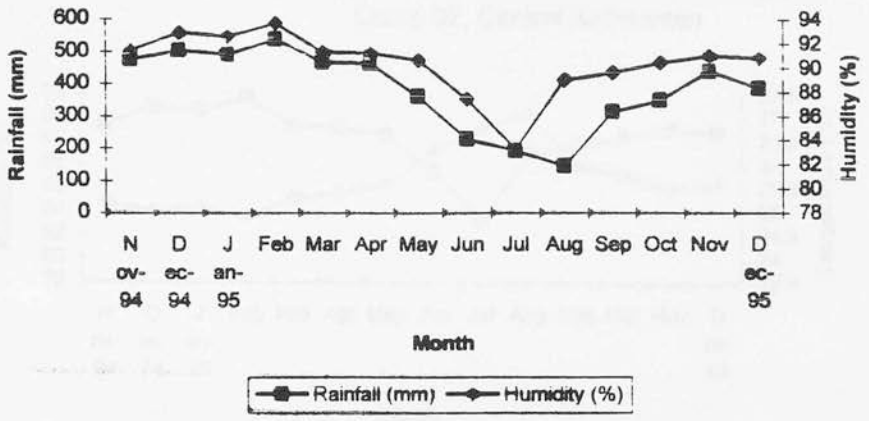


Fig 5.2. Relationship between rainfall and humidity (average) during wet and dry period (1994/1995) in the 2-year-old Logged Plots at Camp 92, Central Kalimantan



The differences between the amount of rain fall in each month during wet and dry seasons at the 2-year logged plots in 1995 was also quite contrast. For example, the amount of rain fall during wet season ranging from 300 to 500 mm but only slightly more than 100 mm during dry season. The humidity was varied between 86-92% during wet season and only around 82% during dry season in the logged plots.

Fig 5.3. Relationship between humidity and temperature (average) during wet and dry period (1994/1995) in the Unlogged Plots at Camp 92, Central Kalimantan

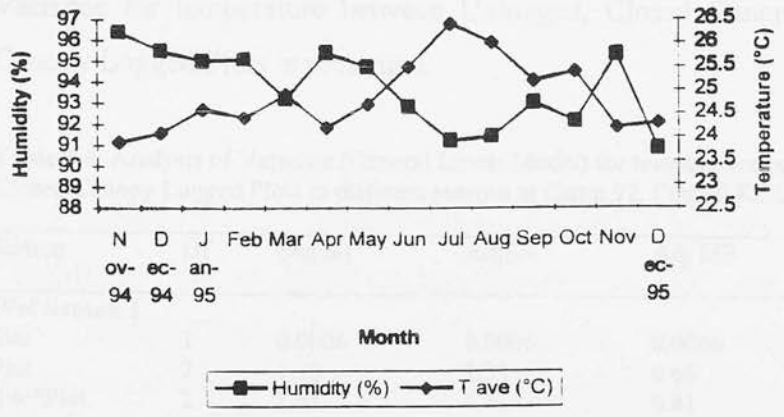


Figure 5.3 shows the differences between temperature in the wet and dry seasons was quite significant. The average temperature was about 23.5°C during wet season and around 25.5°C during dry season in the unlogged plots.

Fig 5.4. Relationship between humidity and temperature (average) during wet and dry period (1994/1995) in the 2-year-old Logged Plots at Camp 92, Central Kalimantan

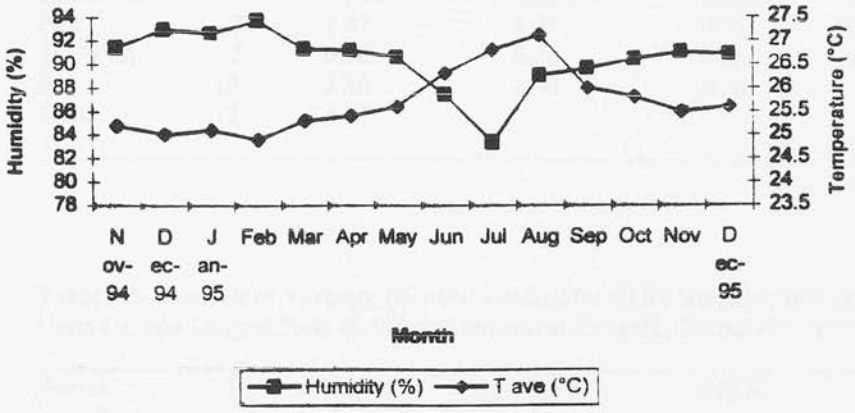


Figure 5.4 shows that the average temperature in the wet season was about 25°C and about 27°C in the dry season in the 2-year old Logged Plots.

Figures 5.3 and 5.4 indicate that the humidity was much lower in the logged plots than in the unlogged plots but in contrast the temperature was much higher in the logged plots in comparison to the unlogged plots. In Tables 5.4 and 5.5 Analysis of Variance for temperature between Unlogged, Closed Canopy Logged and Open Canopy Logged Plots is presented.

Table 5.4. Analysis of Variance (General Linear Model) for temperature between Unlogged and Closed Canopy Logged Plots in different seasons at Camp 92, Central Kalimantan

Source	Df	Seq SS	AdjSS	Adj MS	F	P
Wet Season 1						
Site	1	0.0006	0.0006	0.0006	0.03	0.966
Plot	2	1.33	1.33	0.66	2.25	0.148
Site*Plot	2	1.61	1.61	0.81	2.73	0.105
Error	12	3.55	3.55	0.29		
Total	17	6.50				
Wet Season 2						
Site	1	0.009	0.009	0.009	0.02	0.896
Plot	2	0.27	0.27	0.13	0.27	0.765
Site*Plot	2	0.78	0.78	0.39	0.78	0.480
Error	12	6.00	6.00	0.50		
Total	17	7.07				

Table 5.4 ....contd.

Source	Df	Seq SS	AdjSS	Adj MS	F	P
Dry Season						
Site	1	0.014	0.014	0.014	0.07	0.801
Plot	2	1.47	1.47	0.73	3.53	0.062
Site*Plot	2	0.56	0.56	0.28	1.35	0.295
Error	12	2.50	2.50	0.20		
Total	17	4.55				

Table 5.5. Analysis of Variance (General Linear Model) for temperature between Unlogged and Open Canopy Logged Plots in different seasons at Camp92, Central Kalimantan

Source	Df	Seq SS	AdjSS	Adj MS	F	P
Wet Season 1						
Site	1	6.72	7.42	7.42	29.06	0.000***
Plot	2	1.90	2.09	1.04	4.10	0.144
Site*Plot	2	0.75	0.75	0.37	1.48	0.267
Error	12	3.06	3.06	0.25		
Total	17	6.50				
Wet Season 2						
Site	1	8.22	8.03	8.03	20.23	0.000***
Plot	2	0.03	0.008	0.004	0.01	0.989
Site*Plot	2	1.02	1.02	0.51	1.29	0.311
Error	12	4.76	4.76	0.39		
Total	17	14.04				
Dry Season						
Site	1	0.87	0.91	0.91	5.37	0.003**
Plot	2	0.77	0.72	0.36	2.13	0.162
Site*Plot	2	1.42	1.42	0.71	4.17	0.142
Error	12	2.05	2.05	0.17		
Total	17	5.12				

The results of the GLM test (Table 5.4 and 5.5) indicate that there were no significant differences between temperature in the Unlogged Plots in comparison to the Logged Closed Canopy Plots. However, highly significant differences in temperature were found between the Unlogged and Open Canopy Logged Plots (GLM ANOVA, $P < 0.001$).

Table 5.6. Analysis of Variance (General Linear Model) for humidity between Unlogged and Closed Canopy Logged Plots in different seasons at Camp 92, Central Kalimantan

Source	Df	Seq SS	AdjSS	Adj MS	F	P
Wet Season 1						
Site	1	0.02	0.02	0.02	0.04	0.846
Plot	2	1.05	1.05	0.52	0.76	0.490
Site*Plot	2	3.18	3.18	1.59	2.29	0.144
Error	12	8.34	8.34	0.69		
Total	17	12.60				
Wet Season 2						
Site	1	0.37	0.37	0.37	0.62	0.447
Plot	2	0.49	0.49	0.24	0.41	0.676
Site*Plot	2	2.36	2.36	1.18	1.94	0.186
Error	12	7.30	7.30	0.61		
Total	17	10.54				
Dry Season						
Site	1	0.40	0.40	0.40	0.09	0.774
Plot	2	116.81	116.81	58.41	12.44	0.010**
Site*Plot	2	10.72	10.72	5.36	1.14	0.352
Error	12	56.34	56.34	4.69		
Total	17	184.27				

Table 5.7. Analysis of Variance (General Linear Model) for humidity between Unlogged and Open Canopy Logged Plots in different seasons at Camp 92

Source	Df	Seq SS	AdjSS	Adj MS	F	P
Wet Season 1						
Site	1	23.80	23.80	23.80	19.96	0.001***
Plot	2	3.76	3.76	1.88	1.58	0.246
Site*Plot	2	1.34	1.34	0.67	0.56	0.584
Error	12	14.31	14.31	1.19		
Total	17	43.22				
Wet Season 2						
Site	1	23.57	23.57	23.57	52.65	0.000***
Plot	2	0.61	0.61	0.30	0.68	0.525
Site*Plot	2	2.60	2.60	1.30	2.90	0.094
Error	12	5.37	5.37	0.44		
Total	17	32.16				
Dry Season						
Site	1	113.50	113.50	113.50	81.82	0.000**
Plot	2	32.76	32.76	16.38	11.81	0.010**
Site*Plot	2	68.99	68.99	34.49	24.87	0.100
Error	12	16.64	16.64	1.38		
Total	17	231.89				

There was also no significant difference in humidity between Unlogged Plots and the Closed Canopy Logged Plots (Table 5.6). In contrast, the differences between humidity in the Unlogged Plots and the Open Canopy Logged plots were highly significant (GLM ANOVA, $P < 0.001$) (Table 5.7). There was also a significant difference in humidity between plots in the dry season.

5.3. Discussion

The forest microclimate is influenced by the presence of the canopy and is characterised by a reduction in light levels, air temperature and increased humidity (Ashton 1992, Brown 1993). The physical environment within a forest is significantly altered by logging activities. The energy balance of the forest is altered because of increased penetration of radiation into the lower levels of the canopy and to the forest floor. This in turn affects other aspects of the physical environment including air and soil temperature, humidity and the water balance of the forest (Ashton 1992). Brown (1993) also stated that the change in energy balance of the forest in gaps following logging leads to increased temperatures and decreased relative humidity. In this study, forest conditions in the unlogged plots were similar to these in the closed canopy logged plots (see Chapter III), so, there was no significant differences in humidity and temperature. However, there were significant differences in humidity and temperature compared to the open canopy logged plots because of the presence of large canopy gaps. This is likely to affect animals such as amphibians, which depend on high humidity for survival. The existence of a pronounced dry season will exacerbate the effects of decreased humidity, in more open areas.

Logging reduces the canopy cover and creates artificial canopy gaps with a variety of sizes and shapes that depend on the method and intensity of logging (Whitmore 1993). The difference in percent canopy cover between unlogged plots and logged plots with mechanized selective logging in West Kalimantan was reported to be 40% (Cannon *et al.*1994). This is in line with the results of this study, in which selective logging resulted in heavy disturbance to 45% of the forest canopy. The

mean gap size was clearly larger in logged than in unlogged forest because artificial gaps all came from the largest, fully crowned trees and were often enlarged by dragging manoeuvres, whilst many natural gaps were created by the fall of dead trunks. Skorupa (1988) also recorded 20 and 40% decreases in canopy cover in lightly and heavily logged forest respectively in Kibale Forest, Uganda.

Schmitt (1990) stated that in total about 30% of vegetation on the forest floor, especially the forest litter, in west Bornean lowland dipterocarp forest was disrupted by roads, tractor tracks and skid trails. Damaged areas were characterized by a reduction of forest litter cover. The results of this study also show that the percent litter cover was much higher in the unlogged plots compared to the logged plots, and this suggests that populations of litter fauna, including amphibians, may be adversely affected by logging.

CHAPTER VI
PITFALL TRAPPING

6.1. Introduction

Amphibians are subjects of less than 10 percent of recent studies in ecology and wildlife ecology (Gibbons 1988), yet they are numerically dominant in many habitats and supply an important component of the energy present in terrestrial and aquatic ecosystems. Although the number of species of amphibians is low relative to mammals and birds (Harris and Maser 1984), they are a distinctive and important component of vertebrate fauna (Bury 1988).

There are approximately 134 species of anurans (the collective name for frogs and toads) found in the North Bornean tropical rain forest (Inger and Stuebing 1994). Species adapted to a specific primary forest habitat might be expected to be sensitive to a major disturbance of that habitat (logging). Studies (by pitfall trapping) in northern Kalimantan found significant differences between Unlogged stands and the Logged Stands of Dipterocarp forest in species composition and abundance of anurans (Achmad 1982). The present study results are in line with the northern Kalimantan studies.

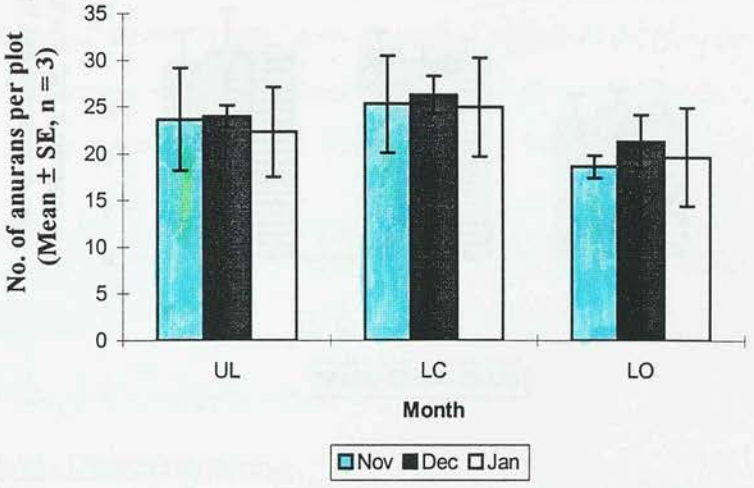
In this chapter, the results of sampling amphibians by pitfall trapping are presented in several sections ie. abundance of anurans, species richness and composition, and species overlap. Microclimate and vegetation components in relation to abundance of anurans and their species richness are also discussed.

6.2. Abundance of anurans

6.2.1. *Differences in abundance*

The average number of anurans captured per study plot showed a small variation between seasons (Figure 6.1, 6.2 and 6.3) in particular for the Unlogged and Closed Canopy Logged Plots. The average number of anurans in the Open Canopy Logged Plot was much lower in comparison to the Unlogged and Logged Closed Plots.

Figure 6.1. Number of individual anurans in different sites during wet season 1 at Camp 92



Bars are standard errors UL = Unlogged Plot LC = Closed Canopy of Logged Plot
LO = Open Canopy of Logged Plot

Figure 6.2. Number of individual anurans in different sites during wet season 2 at Camp 92

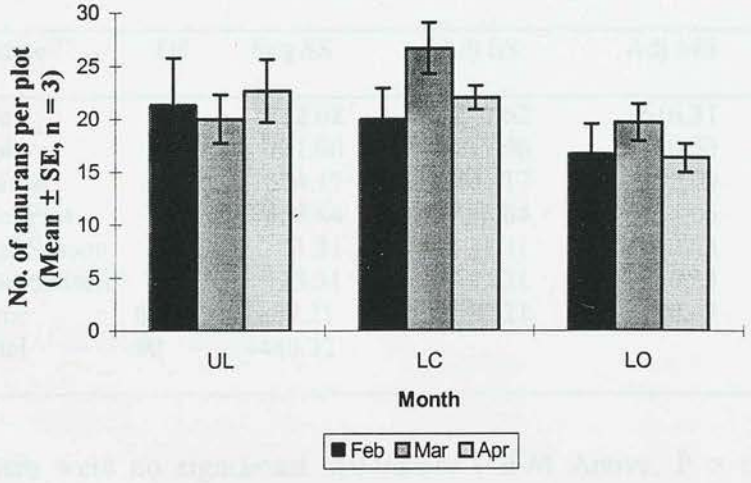
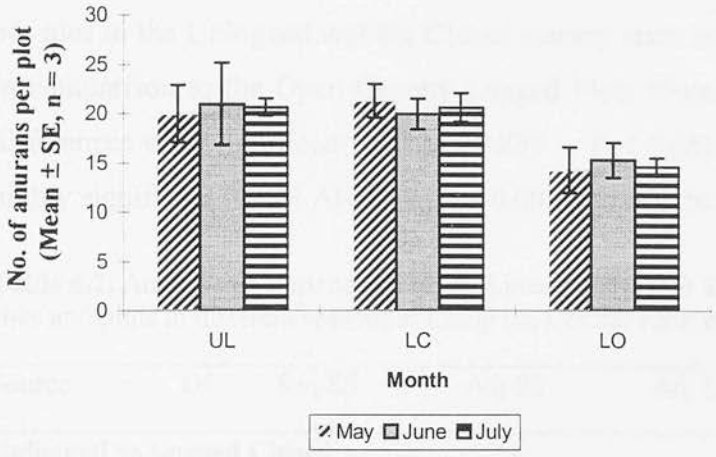


Figure 6.3. Number of individual anurans in different sites during the dry season at Camp



Analysis of Variance (General Linear Model) showed that there were significant differences between the mean number of anurans between sites and between the plots. However, there was no significant difference in abundance of anurans between seasons. (Table 6.1).

Table 6.1. Analysis of Variance (General Linear Model) for Abundance of anurans between sites, plots and seasons at Camp 92, Central Kalimantan

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Site	2	1232.62	1232.62	616.31	20.23	0.000***
Plot	2	221.06	221.96	110.53	3.63	0.032*
Season	2	504.17	504.17	252.09	8.27	0.101
Site*Plot	4	458.64	458.64	114.66	3.76	0.108
Site*Season	4	11.31	11.31	2.83	0.09	0.984
Plot*Season	4	123.31	123.31	30.83	1.01	0.408
Error	62	1889.21	1889.21	30.47		
Total	80	4440.32				

There were no significant differences (GLM Anova, $P > 0.05$) in abundance of anurans between Unlogged and Closed Canopy Logged Sites either in dry or wet period as indicated in Table 6.2. This is possibly due to the similarity of forest conditions (composition, structure, and vegetation cover) and microclimate (i.e. humidity and temperature) of both forests. However, there was a significant difference on abundance of anurans in the observed plot during the dry season

(Table 6.2). This is likely due to the variation of dry conditions between plots that affects the abundance of anurans. The mean numbers of individual anurans captured per plot in the Unlogged and the Closed canopy areas of Logged Plots were higher in comparison to the Open Canopy Logged Plots (Figure 6.1, 6.2 and 6.3). These differences were significant (GLM ANOVA, $P < 0.05$) during the wet season and highly significant (GLM ANOVA, $P < 0.001$) during the dry season (Table 6.2).

Table 6.2. Analysis of Variance (General Linear Model) for abundance of anurans between sites and plots in different seasons at Camp 92, Central Kalimantan

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Unlogged vs Logged Closed						
Wet I (November-January)						
Plot	2	165.78	165.78	82.89	1.30	0.309
Site	1	5.56	5.56	5.56	0.09	0.773
Plot*Site	2	311.11	311.11	155.56	2.43	0.130
Error	12	766.67	766.67	63.89		
Total	17	1249.11				
Wet II (February-April)						
Plot	2	56.44	56.44	28.22	0.86	0.447
Site	1	2.00	2.00	2.00	0.06	0.809
Plot*Site	2	148.00	148.00	74.00	2.26	0.147
Error	12	392.67	392.67	32.72		
Total	17	599.11				
Dry (May-July)						
Plot	2	176.78	176.78	88.39	7.50	0.008**
Site	1	0.00	0.00	0.00	0.00	1.000
Plot*Site	2	19.00	19.00	9.50	0.81	0.469
Error	12	141.33	141.33	11.78		
Total	17	337.11				
Unlogged vs Logged Open						
Wet I (November-January)						
Plot	2	141.56	141.56	70.78	1.29	0.299
Site	1	646.22	646.22	323.11	5.91	0.011**
Plot*Site	2	343.56	343.56	85.89	1.57	0.225
Error	12	984.67	984.67	54.70		
Total	17	2116.00				
Wet II (February-April)						
Plot	2	29.56	29.56	14.78	0.57	0.575
Site	1	354.67	354.67	177.33	6.86	0.006**
Plot*Site	2	256.44	256.44	64.11	2.48	0.081
Error	12	465.33	465.33	25.85		
Total	17	1106.00				

Table ....contd.

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Dry (May-July)						
Plot	2	101.40	101.40	50.70	5.48	0.014*
Site	1	373.40	373.40	186.70	20.16	0.000***
Plot*Site	2	103.03	103.03	25.76	2.78	0.06
Error	12	166.66	166.66	9.26		
Total	17	744.52				
Logged Closed vs Logged Open						
Wet I (November-January)						
Plot	2	112.33	112.33	56.17	1.24	0.324
Site	1	430.22	430.22	430.22	9.50	0.009**
Plot*Site	2	52.11	52.11	26.06	0.58	0.557
Error	12	543.33	543.33	45.28		
Total	17	1138.00				
Wet II (February-April)						
Plot	2	70.78	70.78	35.39	1.85	0.200
Site	1	242.00	242.00	242.00	12.63	0.004**
Plot*Site	2	22.33	22.33	11.17	0.58	0.573
Error	12	230.00	230.00	19.17		
Total	17	565.11				
Dry (May-July)						
Plot	2	15.44	15.44	7.72	0.77	0.485
Site	1	280.06	280.06	280.06	27.85	0.000***
Plot*Site	2	33.44	33.44	16.72	1.66	0.230
Error	12	120.67	120.67	10.06		
Total	17	449.61				

* ($P < 0.05$) ** ($P < 0.01$) *** ($P<0.001$)

Table 6.2 showed that there was a significant difference in abundance of anurans between plots which was only observed during the dry season. This could be an effect of drying conditions during the dry season. In the dry season the humidity become patchy and low whilst the temperature is higher (as mentioned earlier in the Chapter V) than in the wet season. Anurans have a low tolerance for hot and dry conditions, therefore their distribution pattern becomes more clumpy and they tend to distribute to the wetter places (ie. under logs, leaf litter) which are important sources of moisture throughout the season. The significant differences in abundance of anurans during the dry season may reflect unfavorable environmental conditions, probably related to moisture stress.

6.2.2. *Microclimate and abundance of anurans*

Rainfall, humidity and temperature had significant effects (ANCOVA, $P < 0.001$) on the abundance of anurans in this study (Table 6.3, 6.4 and 6.5).

Table 6.3. Analysis of Covariance for abundance of anurans in relation to rainfall between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet Season 1					
Rainfall	3	811.59	270.53	17.92	0.000***
Site	2	64.62	32.31	2.14	0.152
Plot	2	10.43	5.21	0.35	0.713
Site*Plot	4	95.75	23.94	1.59	0.229
Error	15	226.41	15.09		
Total	26	2085.63			
Wet Season 2					
Rainfall	3	365.86	121.95	18.39	0.000***
Site	2	60.14	30.07	4.53	0.029*
Plot	2	15.73	7.86	1.19	0.333
Site*Plot	4	72.77	18.19	2.74	0.068
Error	15	99.47	6.63		
Total	26	1106.00			
Dry Season					
Rainfall	3	69.99	23.33	3.62	0.003**
Site	2	260.23	130.11	20.19	0.000***
Plot	2	19.69	9.84	1.53	0.249
Site*Plot	4	13.22	3.30	0.51	0.727
Error	15	96.67	6.44		
Total	26	744.52			

Table 6.4. Analysis of Covariance for abundance of anurans in relation to air humidity between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet Season 1					
Humidity	3	950.57	316.86	54.36	0.000***
Site	2	86.60	43.30	7.43	0.006**
Plot	2	5.21	2.61	0.45	0.648
Site*Plot	4	48.76	12.19	2.09	0.133
Error	15	87.43	5.83		
Total	26	2085.63			
Wet Season 2					
Humidity	3	369.34	123.11	19.24	0.000***
Site	2	70.76	35.38	5.53	0.016**
Plot	2	26.09	13.04	2.04	0.165
Site*Plot	4	82.08	20.52	3.21	0.093
Error	15	95.99	6.40		
Total	26	1106.00			

Table ...contd.

Source	Df	Adj SS	MS	F	P
Dry Season					
Humidity	3	127.08	42.36	16.05	0.000**
Site	2	8.75	4.38	1.66	0.023**
Plot	2	17.67	8.83	3.35	0.063
Site*Plot	4	3.24	0.81	0.31	0.869
Error	15	39.58	2.64		
Total	26	744.52			

Table 6.5. Analysis of Covariance for abundance of anurans in relation to air temperature between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet Season 1					
Temperature	3	755.45	251.82	13.37	0.000***
Site	2	21.11	10.55	0.56	0.583
Plot	2	17.57	8.78	0.47	0.636
Site*Plot	4	83.31	20.83	1.11	0.390
Error	15	282.55	18.84		
Total	26	2085.63			
Wet Season 2					
Temperature	3	360.12	120.04	17.11	0.000***
Site	2	4.78	2.39	0.34	0.716
Plot	2	69.85	34.92	4.98	0.022*
Site*Plot	4	70.48	17.62	2.51	0.086
Error	15	105.21	7.01		
Total	26	1106.00			
Dry Season					
Temperature	3	125.11	41.70	15.06	0.000**
Site	2	22.10	11.05	3.99	0.041*
Plot	2	19.53	9.76	3.53	0.046*
Site*Plot	4	14.72	3.68	1.33	0.304
Error	15	41.55	2.77		
Total	26	744.52			

Rainfall as well as humidity had significant effects on the abundance of amphibians because they are strongly influence anuran activity, distribution and dispersion patterns. The combination of air temperature and humidity determines the rate of water loss from anuran moist skin. Therefore, it is also affect the distribution and activity patterns of the anurans.

Linear Regression was used to examine the relationship between humidity and abundance of anurans during dry and wet seasons between sites (Figure 6.4, 6.5 and 6.6) and between temperature and the abundance of anurans, respectively (Figure 6.7, 6.8 and 6.9).

Fig. 6.4. Regression analysis for humidity and abundance of anurans during wet season 1 at Camp 92, Central Kalimantan

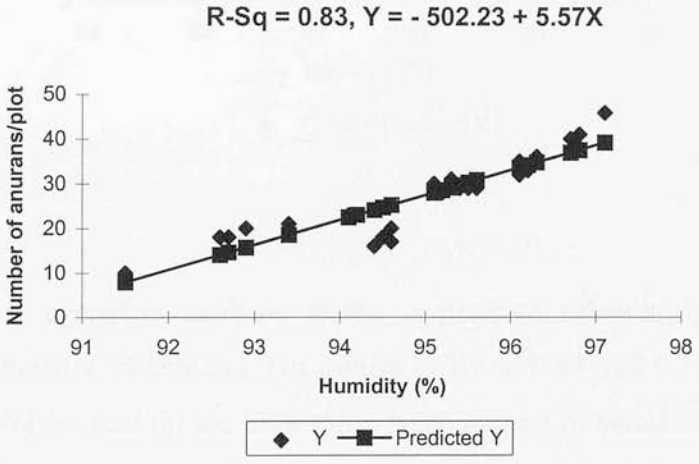


Fig. 6.5. Regression analysis for humidity and abundance of anurans during wet season 2 at Camp 92, Central Kalimantan

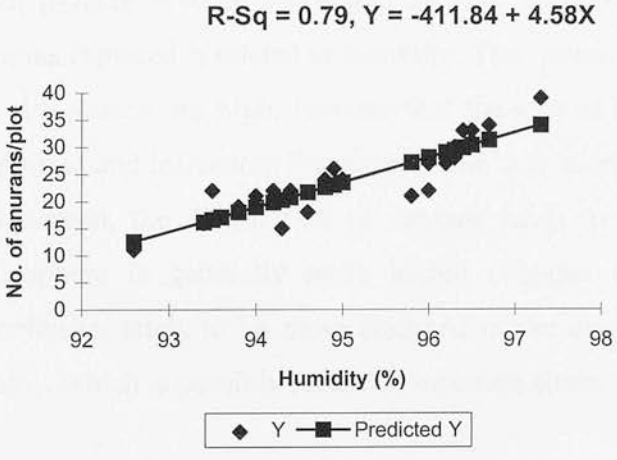
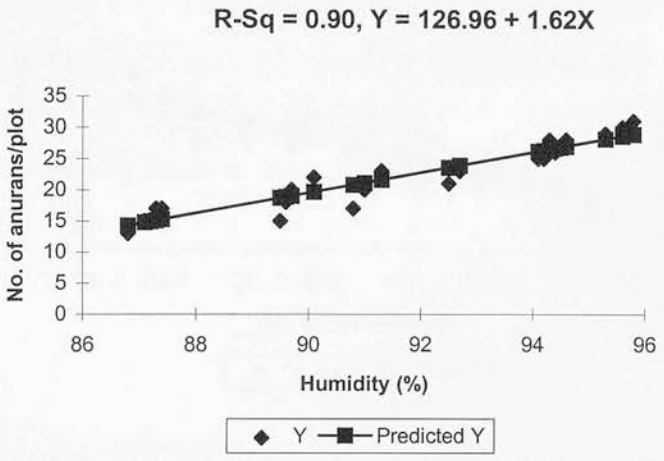


Fig. 6.6. Regression analysis for humidity and abundance of anurans during the dry season at Camp 92, Central Kalimantan



The regression analysis shows a positive relationship between humidity and abundance of anurans. The values of R^2 at 0.83 and 0.79 (wet season1 and 2) and 0.90 (dry season) are high and a large amount of variability in the data set has been accounted for by the regression model. A high R^2 indicates that the dependent variable (humidity) explains a high proportion of the variance in the dependent variable (abundance of anurans). The statistical evidence (R-sq = 0.83, 0.79 and 0.90, p-value < 0.0001, n = 27) strongly supports the idea that the number of anurans captured is related to humidity. The values of R^2 in wet seasons 1 and 2 and the dry season are high, indicate that the role of humidity at any seasons is very important and influences the survivorship and activity pattern of the anurans. In the wet season, the distribution of anurans tends to be more dispersed because the atmosphere is generally more humid (Figures 6.4 and 6.5). However, their distribution tends to be more clumped in the dry season because the humidity is patchy, which is possibly related to moisture stress.

Fig. 6.7 Regression analysis for air temperature and abundance of anurans during wet season 1 at Camp 92, Central Kalimantan

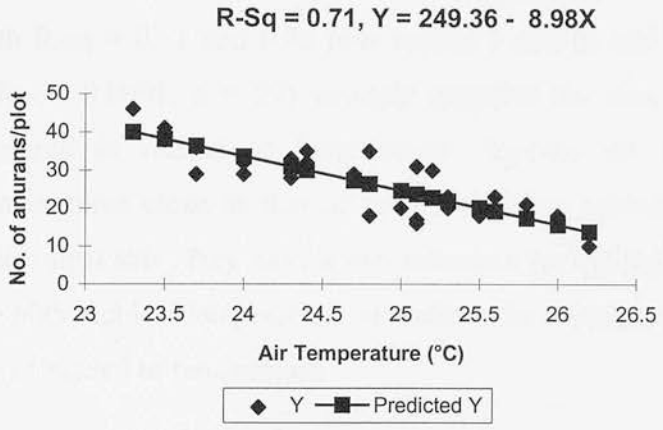


Fig. 6.8. Regression analysis for air temperature and abundance of anurans during wet season 2 at Camp 92, Central Kalimantan

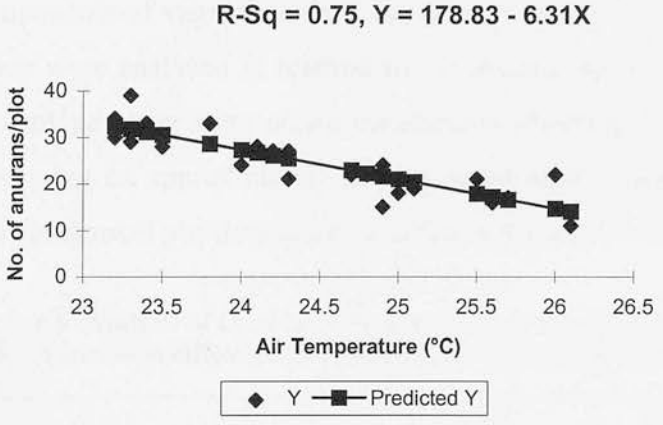
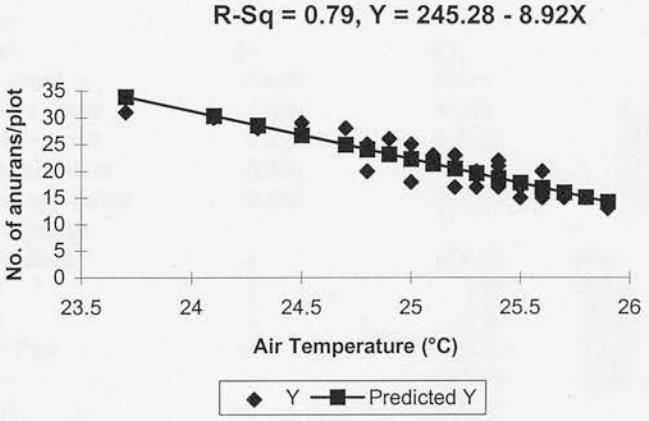


Fig. 6.9. Regression analysis for temperature and abundance of anurans during the dry season at Camp 92, Central Kalimantan



The regression analysis shows a negative relationship in which increase of temperature leads to a decrease in the abundance of anurans.The statistical evidence with R-sq = 0.71 and 0.75 (wet season 1 and 2) and R-sq = 0.79 (dry season), p-value < 0.0001, n = 27) strongly supports the idea that the number of anurans captured is related to temperature. Anurans are ectotherms and have body temperatures close to that of their immediate surroundings. However because of their moist skin, they have a low tolerance for hot and dry conditions especially to the high ambient temperature. Therefore, the abundance of anurans was decreased in the increased of temperature.

6.2.3. *Vegetation components and abundance of anurans*

Vegetation can be used as a basis for specific site descriptions and several components of vegetation such as canopy cover, shrub cover, herb cover, and litter cover were analysed in relation to the abundance of anurans. Variables measuring percentage cover were arcsin-transformed (Sokal and Rohlf 1995) before analysis, producing an approximately normal distribution. Associations between abundance and categorical physiographic variables were tested with ANCOVA.

Table 6.6 Analysis of Covariance for abundance of anurans in relation to % plant and litter cover between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet Season					
Covariates	4	946.20	236.55	36.08	0.000***
Site	2	38.45	19.22	2.93	0.086
Plot	2	58.41	29.21	4.45	0.132
Site*Plot	4	75.43	18.86	2.88	0.062
Error	14	91.80	6.56		
Total	26	2085.63			
Covariate	Coeff	Stdev	t-value	P	
% litter cover	2.304	0.198	11.623	0.000***	
% herb cover	0.184	0.238	0.775	0.451	
% shrub cover	0.214	0.333	0.644	0.530	
% canopy cover	0.496	0.461	1.076	0.300	
Dry Season					
Covariates	4	137.12	34.28	16.24	0.000***
Site	2	0.13	0.06	0.03	0.969
Plot	2	2.66	1.33	0.63	0.546
Site*Plot	4	12.62	3.15	1.50	0.257
Error	14	29.55	2.11		
Total	26	744.52			

Table ... contd.

Covariate	Coeff	Stdev	t-value	P
% litter cover	2.207	0.338	6.537	0.000***
% herb cover	0.016	0.174	0.095	0.925
% shrub cover	0.144	0.223	0.646	0.529
% canopy cover	0.092	0.267	1.346	0.735

Table 6.6 indicates that plant cover (ie.herb cover, shrub and canopy cover) has no significant effects on the abundance of anurans (GLM ANOVA, $P > 0.05$). However, percent litter cover has a very significant effect on the abundance of anurans during wet and dry seasons (GLM ANOVA, $P < 0.001$). Regression analysis was done to determine the precise relationship between % litter cover and the abundance of anurans (Figure 6.10 and 6.11).

Fig. 6.10. Regression analysis for % litter cover and abundance of anurans during wet season at Camp 92, Central Kalimantan

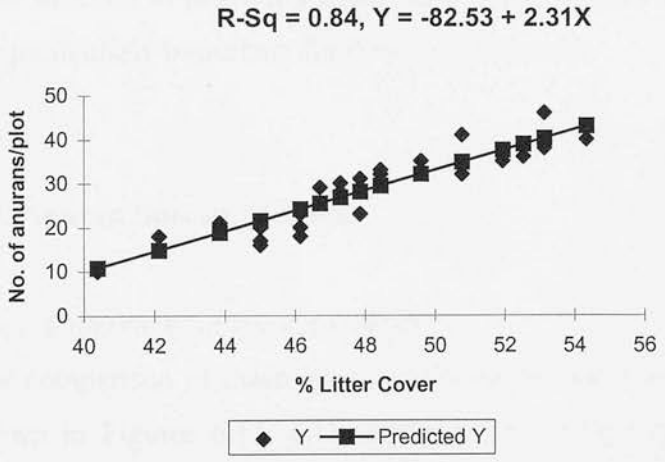
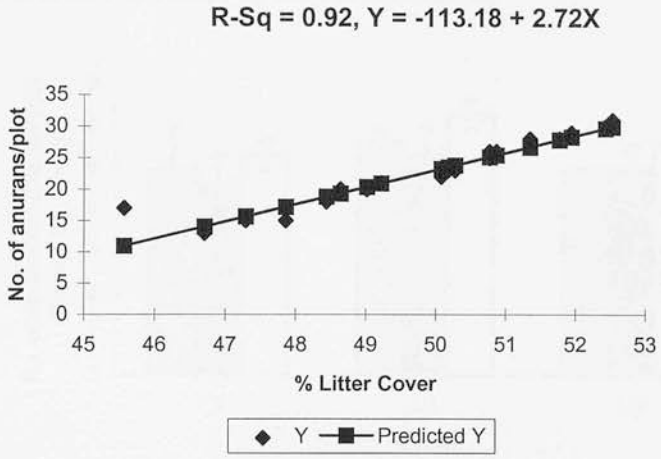


Fig. 6.11. Regression analysis for % litter cover and abundance of anurans during the dry season at Camp 92, Central Kalimantan



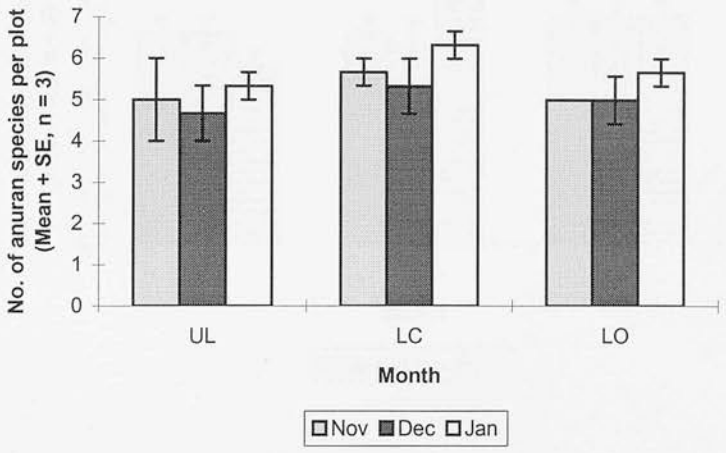
This regression (R-sq = 0.84 and 0.92, p-value < 0.0001, n = 27) indicated that the number of anurans captured was related to % litter cover. It shows a positive relationship between % litter cover and the number of anurans captured. Table 6.6 also shows that there were very highly significant effects of litter cover on abundance of anurans at any seasons. It is likely to suggest that litter cover has a vital function in providing shelter and maintaining the humidity of the soil which are particularly important for survival of anurans.

6.3. Anuran Species Richness

6.3.1. *Differences in species richness*

The comparison of mean species richness per site between wet and dry seasons is shown in Figures 6.12, 6.13, and 6.14. From the figures, the number of anuran species captured per study plot varied slightly between sites in different seasons and the average number of species in the wet season was slightly higher than in the dry season but the differences were not statistically significant.

Figure 6.12. Number of anuran species in different sites captured by pitfall traps during wet season 1 at Camp 92, Central Kalimantan



Bars are standard errors UL = Unlogged Plot LC = Closed Canopy Logged Plot LO = Open Canopy Logged Plot

Figure 6.13. Number of anuran species in different sites captured by pitfall traps during wet season 2 at Camp 92, Central Kalimantan

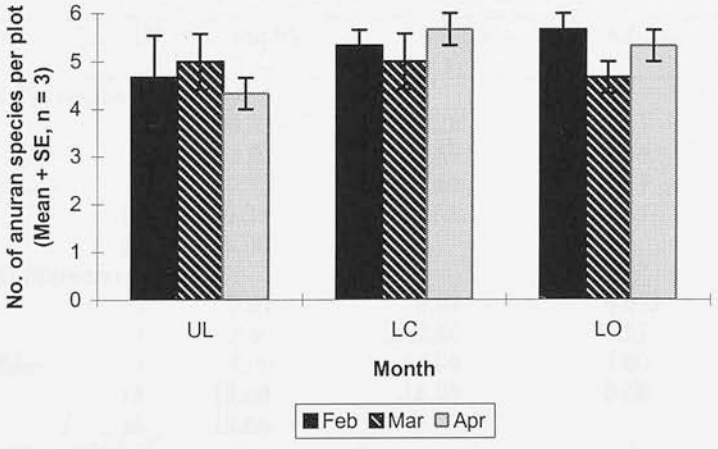
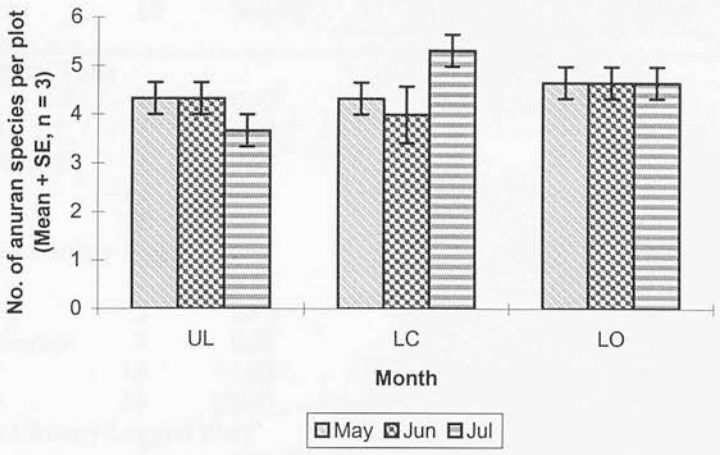


Figure 6.14. Number of anuran species in different sites captured by pitfall traps during dry season at Camp 92, Central Kalimantan



There was no significant difference between the mean number of anuran species per site (GLM ANOVA, $P > 0.05$) (Table 6.7). Neither were there significant differences in species richness between seasons. In other words, no seasonal effects on anuran species richness were observed (Table 6.8)

Table 6.7. Analysis of Variance (General Linear Model), significant test for species richness between sites in different seasons at Camp 92

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Wet I (November-January)						
Plot	2	0.22	0.22	0.11	0.13	0.879
Site	2	2.89	2.89	1.44	1.70	0.212
Plot*Site	4	3.55	3.55	0.89	1.04	0.412
Error	18	15.33	15.33	0.85		
Total	26	22.00				
Wet II (February-April)						
Plot	2	0.01	0.01	0.005	0.03	1.000
Site	2	2.66	2.66	1.33	2.00	0.164
Plot*Site	4	4.00	4.00	1.00	1.50	0.244
Error	18	12.00	12.00	0.66		
Total	26	18.66				
Dry (May-July)						
Plot	2	0.07	0.07	0.04	0.07	0.931
Site	2	1.18	1.18	0.59	1.14	0.341
Plot*Site	4	1.92	0.92	0.48	0.93	0.469
Error	18	9.33	9.33	0.52		
Total	26	12.52				

Table 6.8. Analysis of Variance (General Linear Model), significant test for species richness between seasons in different sites at Camp 92

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Unlogged Plots						
Plot	2	5.63	5.63	2.81	3.45	0.064
Season	2	3.63	3.63	1.81	2.23	0.137
Plot*Season	4	0.59	0.59	0.15	0.18	0.945
Error	18	14.66	14.66	0.81		
Total	26	24.52				
Closed Canopy Logged Plots						
Plot	2	1.55	1.55	0.78	1.00	0.387
Season	2	6.88	6.88	3.44	4.43	0.127
Plot*Season	2	0.22	0.22	0.05	0.07	0.990
Error	18	14.00	14.00	0.78		
Total	26	22.66				
Open Canopy Logged Plots						
Plot	2	1.18	0.18	0.59	1.33	0.29
Season	2	3.18	3.18	1.59	3.58	0.059
Plot*Season	4	0.59	0.59	1.15	0.33	0.852
Error	18	8.00	8.00	0.44		
Total	26	12.96				

6.3.2. *Microclimate and species richness*

Microclimate components such as rainfall, humidity and temperature had no significant effects on species richness (ANCOVA, $P > 0.05$) as indicated in Tables 6.9, 6.10 and 6.11.

Table 6.9. Analysis of Covariance for species richness in relation to rainfall between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	Adj MS	F	P
Wet Season 1					
Rainfall	3	2.64	0.88	1.04	0.402
Site	2	2.93	1.46	1.73	0.210
Plot	2	0.44	0.22	0.27	0.771
Site*Plot	4	4.66	1.16	1.38	0.288
Error	15	12.68	0.84		
Total	26	22.00			
Wet Season 2					
Rainfall	3	0.39	0.13	0.17	0.916
Site	2	1.00	0.50	0.65	0.538
Plot	2	0.04	0.02	0.03	0.971
Site*Plot	4	3.70	0.92	1.20	0.353
Error	15	11.61	0.77		
Total	26	18.66			

Table ... contd.

Source	Df	Adj SS	Adj MS	F	P
Dry Season					
Rainfall	3	2.01	0.67	1.37	0.289
Site	2	0.42	0.21	0.44	0.653
Plot	2	0.27	0.13	0.28	0.762
Site*Plot	4	2.65	0.66	1.36	0.294
Error	15	11.61	0.77		
Total	26	18.66			

Table 6.10. Analysis of Covariance for species richness in relation to humidity between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	Adj MS	F	P
Wet Season 1					
Humidity	3	3.63	1.21	1.55	0.242
Site	2	1.83	0.91	1.17	0.336
Plot	2	0.07	0.03	0.05	0.953
Site*Plot	4	3.13	0.78	1.01	0.435
Error	15	11.69	0.77		
Total	26	22.00			
Wet Season 2					
Humidity	3	0.08	0.02	0.04	0.991
Site	2	1.84	0.92	1.16	0.341
Plot	2	0.10	0.05	0.07	0.998
Site*Plot	4	3.27	0.81	1.03	0.424
Error	15	11.91	0.79		
Total	26	18.66			
Dry Season					
Humidity	3	0.27	0.09	0.15	0.928
Site	2	0.78	0.39	0.64	0.539
Plot	2	0.10	0.05	0.09	0.918
Site*Plot	4	1.52	0.38	0.63	0.647
Error	15	9.06	0.60		
Total	26	12.52			

Table 6.11. Analysis of Covariance for species richness in relation to temperature between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	Adj MS	F	P
Wet Season 1					
Temperature	3	2.39	0.79	0.92	0.453
Site	2	2.57	1.28	1.49	0.257
Plot	2	0.16	0.08	0.09	0.911
Site*Plot	4	4.88	1.22	1.41	0.277
Error	15	12.94	0.86		
Total	26	22.00			

Table ... contd.

Source	Df	Adj SS	Adj MS	F	P
Wet Season 2					
Temperature	3	0.35	0.11	0.15	0.928
Site	2	1.99	0.99	1.29	0.305
Plot	2	0.01	0.008	0.01	0.990
Site*Plot	4	3.68	0.92	1.19	0.357
Error	15	11.64	0.77		
Total	26	18.66			
Dry Season					
Temperature	3	0.70	0.23	0.41	0.748
Site	2	0.95	0.47	0.83	0.455
Plot	2	0.21	0.10	0.19	0.830
Site*Plot	4	1.32	0.33	0.58	0.684
Error	15	8.62	0.57		
Total	26	12.51			

6.3.3. *Vegetation and species richness*

Vegetation variables (shrub cover, herb cover and canopy cover) had no significant effects (GLM ANOVA, $P > 0.05$) on species richness in the wet season (Table 6.12), but litter cover had a significant effect in the dry season ($P < 0.05$).

Table 6.12. Analysis of Covariance for species richness in relation to forest litter cover between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet Season					
Covariates	4	4.12	1.03	1.69	0.209
Site	2	1.67	0.84	1.37	0.285
Plot	2	1.83	0.91	1.50	0.256
Site*Plot	4	0.57	0.14	0.23	0.915
Error	14	8.54	0.61		
Total	26	16.66			
Covariate	Coeff	Stdev	t-value	P	
% litter cover	0.089	0.069	1.283	0.220	
% herb cover	0.034	0.084	0.405	0.691	
% shrub cover	0.143	0.104	1.369	0.193	
% canopy cover	0.239	0.174	1.364	0.193	
Dry Season					
Covariates	4	2.63	0.66	1.72	0.201
Site	2	0.51	0.25	0.67	0.529
Plot	2	0.47	0.24	0.62	0.551
Site*Plot	4	1.50	0.37	0.98	0.448
Error	14	5.36	0.38		
Total	26	9.62			

Table ... contd.

Covariate	Coeff	Stdev	t-value	P
% litter cover	0.137	0.055	2.478	0.027*
% herb cover	0.023	0.068	0.333	0.743
% shrub cover	0.053	0.075	0.710	0.489
% canopy cover	0.039	0.102	0.384	0.706

6.4. Anuran Species Composition

A total of only 12 species was recorded in 3 sites (Table 6.13), probably representing only a small proportion of all Bornean anurans in the study areas. Of these, three species (*Kalophrynus pleurostigma*, *Chaperina fusca* and *Kaloula smythi*, family *Microhylidae*) were microhylids, two were pelobatids (*Leptobrachium abotti* and *Megophrys nasuta*, family *Pelobatidae*), three were bufonids (*Bufo biporcatus*, *Ansonia longidigita*, and *Pedostibes rugosus*, family *Bufonidae*) and four were ranids (*Rana finchi*, *R. chalconota*, *R. palavanensis* and *R. limnocharis*, family *Ranidae*).

Table 6.13. Number of captures of each species from three sites in different seasons at Camp 92, Central Kalimantan

Species/Season	Wet 1			Wet 2			Dry		
(Site)	UL	LC	LO	UL	LC	LO	UL	LC	LO
Leptobrachium abotti	9	9	0	9	9	0	9	9	0
Kalophrynus pleurostigma	8	9	9	9	9	9	9	9	9
Bufo biporcatus	9	9	9	9	9	9	8	9	9
Megophrys nasuta	3	7	1	4	9	1	2	6	0
Chaperina fusca	6	5	0	7	6	0	7	5	1
Ansonia longidigita	3	4	8	2	1	9	0	0	3
Rana finchi	2	2	0	0	0	0	0	1	0
R. chalconota	2	3	3	2	3	5	1	0	4
R. palavanensis	2	2	0	0	0	0	1	0	0
R. limnocharis	0	0	9	0	0	9	0	0	9
Pedostibes rugosus	0	0	8	0	0	6	0	0	8
Kalaoula smythi	0	2	0	0	2	0	0	3	0
Total	45	52	47	42	48	48	37	42	43

UL = Unlogged Plot LC = Closed Canopy Logged Plot LO = Open Canopy Logged Plot

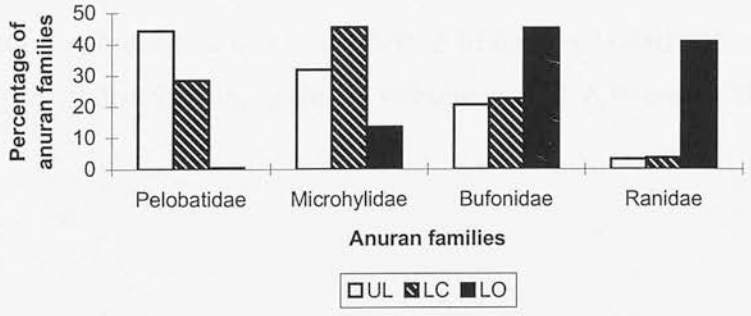
Totals of 9, 10, and 7 species of anurans were captured by pitfall traps during Wet Season 1 in the Unlogged Plots, Closed canopy Logged Plots, and Open canopy Logged Plots, respectively. Totals of 7, 8, and 7 (wet season II) and 7, 7 and 7 (dry

season) species of anurans were also captured at the three different sites (Table 6.9).

Certain species appear to be characteristic of particular forest types. For example, *Leptobrachium abotti* found only in the Unlogged and Closed Canopy Logged Plots. In contrast, *Rana limnocharis* and *Pedostibes rugosus* found only in the Open Canopy Logged Plot.

The proportions of forest-dwelling anurans grouped by families, recorded from pitfall traps and drift fences are shown in Figures 6.15, 6.16 and 6.17. Of the total number of species captured in the unlogged plots, 46.3% of them were represented by pelobatids, 35.3% microhylids, 16.6% bufonids, and the smallest proportion (1.8%) were ranids (*Ranidae*). In the closed canopy areas of logged plots, 46.6% were microhylids, 35.0% pelobatids, 16.7% bufonids and the rest were ranids. In contrast, the largest proportion of anurans captured by pitfalls in open canopy logged plots were bufonids (46.6%, of which around 40% were *Bufo biporcatus*) and Ranids (41.5%, of which around 38% were *Rana limnocharis*). 11.5% consisted of microhylids and the smallest proportion was pelobatids.

Fig 6.15 The proportion of individual anurans in each family captured by pitfall traps in different sites during wet season I at Camp 92, Central Borneo



UL = Unlogged Plot LC = Closed Canopy Logged Plot LO = Open Canopy Logged Plot

Fig 6.16 The proportion of individual anurans in each family captured by pitfall traps in different sites during wet season 2 at Camp 92, Central Borneo

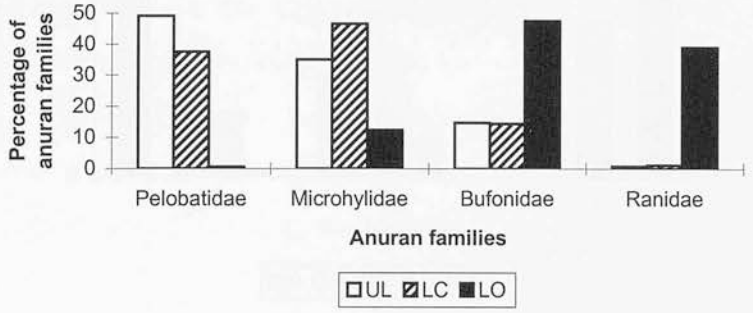
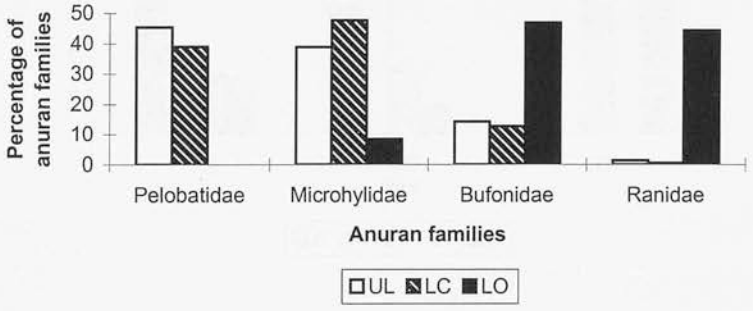
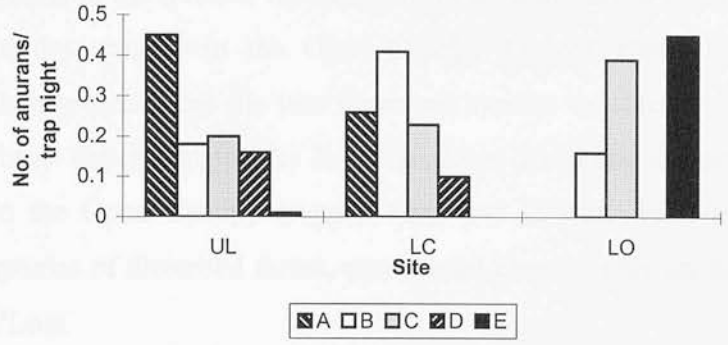


Fig 6.17 A histogram showing the proportion of the frog family captured by pitfall traps in different sites during dry season at Camp 92, Central Borneo



Species composition can be expressed in terms of relative density and graphed as a frequency distribution, as shown in Figures 6.18, 6.19 and 6.20.

Fig 6.18 Frequency distribution of dominant species during wet season 1 in different sites at Camp 92, Central Kalimantan



A = *Leptobrachium abotti* B = *Kalophrynus pleurostigma* C = *Bufo biporcatus* D = *Chaperina fusca* E = *Rana limnocharis*

Fig 6.19 Frequency distribution of dominant species during wet season 2 in different sites at Camp 92, Central Kalimantan

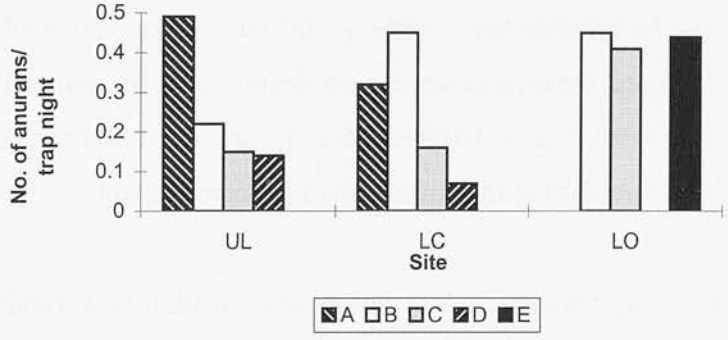
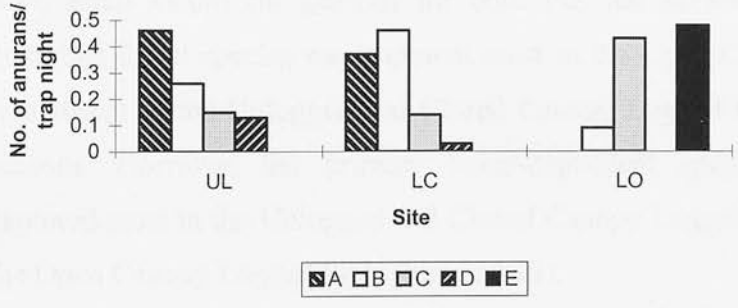


Fig 6.20 Frequency distribution of dominant species during dry season in different sites at Camp 92, Central Kalimantan



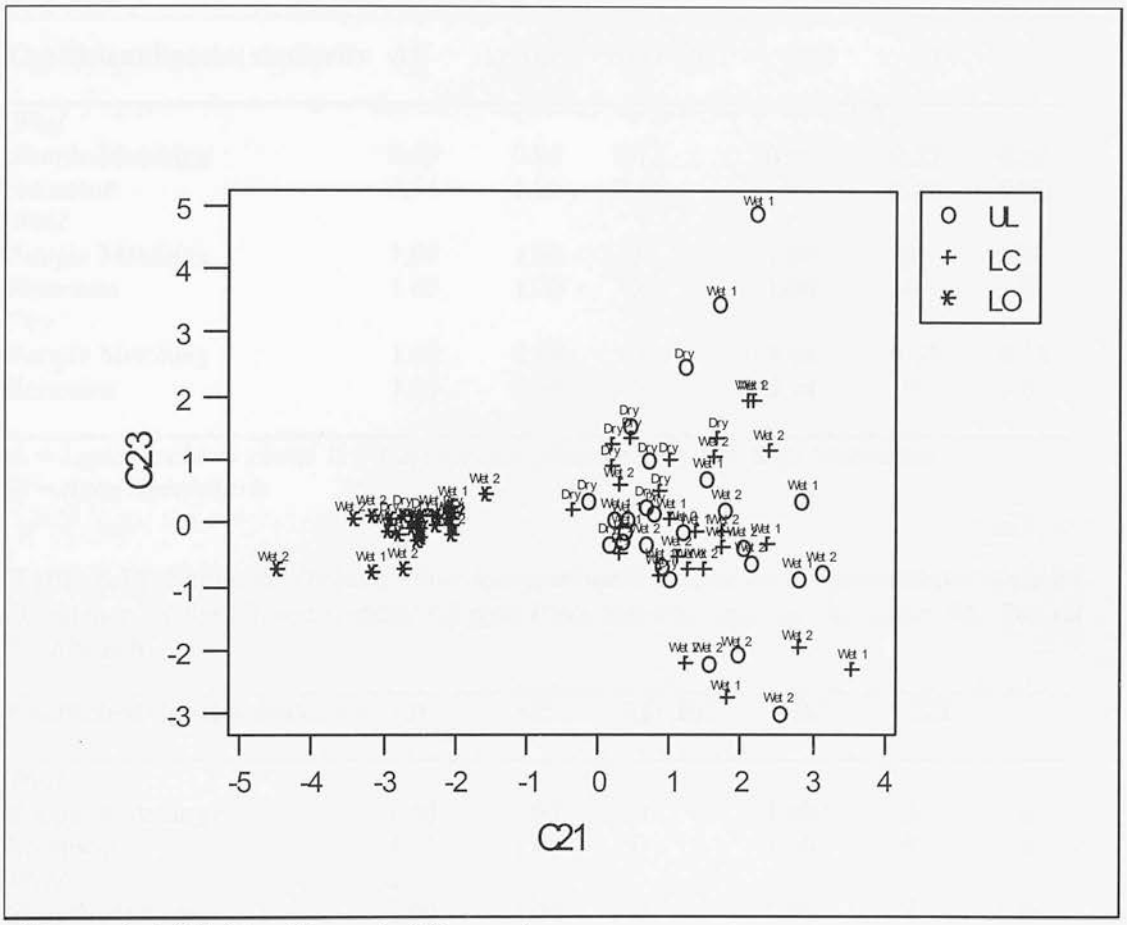
Leptobrachium abotti was the most abundant species captured in the Unlogged, but Kalophrynus seemed to be the most abundant in the Closed Canopy Logged Plots. In addition, the number of individuals captured during the wet season was higher than in dry season. In the Open Canopy Logged Plots, *Bufo biporcatus* and *Rana limnocharis* were the two dominant species captured by pitfall traps. The primary forest dependent species *Leptobrachium abotti* and *Chaperina fusca* were not found in the Open Canopy Logged Plots and in turn, *Rana limnocharis* as the pioneer species of disturbed forest, was almost absent in the primary forest of the Unlogged PLots.

6.5. Presence-Absence of Different Species

Presence-absence data for species in the study sites provided information on species interactions that may be particularly useful in explaining observed distributions. Principal Component Analysis was used to describe the relationship between observed variables and the presence and absence of the species in the study areas. Measures of Association were used to express faunal resemblance as similarity or dissimilarity among species observed at the sample sites and also to assess the degree of coexistence in different sites of particular species of amphibians.

Figure 6.21 indicates that points at the extreme right-hand end of axis 1, correspond to individuals (for example *Leptobrachium abotti* between point +3 and +4) that have large values (in general) for both season and site in the Unlogged and Logged Closed Canopy Plots; while points at the extreme left-hand end of axis 1 correspond to individuals (for example *Kalophrynus pleurostigma* between point -2 and -3) that have small values (in general) for both site and season. *Rana limnocharis* the disturbed forest species was captured most in the Open Canopy Logged Plots and was absent in the Unlogged and Closed Canopy Logged Plots during dry and wet seasons. However, the primary forest-dependent species *Leptobrachium* was captured most in the Unlogged and Closed Canopy Logged Plots and was absent in the Open Canopy Logged Plots (Figure 6.21).

Fig 6.21. PCA chart for presence-absence of species (pitfall traps) between seasons in different sites at Camp 92, Central Kalimantan



c21 = axis 1 (site) c23 = axis 2 (season)

6.5. Species similarity

The similarity between species occurrence in the sampling areas was approached by the coefficient of association. The coefficient of association was calculated based on the abundance of anurans data. These coefficients reflect the proportion of the sample that represents mutual occurrence (Hohn 1976). Measures, or coefficients, of association are used to assess the degree of coexistence in different localities or habitats of particular species of amphibians. (Heyer *et al.* 1994).

Table 6.14. Similarities between four dominant species based on the proportion of anurans abundance in the unlogged plots between seasons at Camp 92, Central Kalimantan

Coefficient/Species similarity	AB	AC	AD	BC	BD	CD
Wet1						
Simple Matching	0.89	0.90	0.11	0.89	0.22	0.20
Sorenson	0.94	1.00	0.20	0.94	0.22	0.20
Wet2						
Simple Matching	1.00	1.00	0	1.00	0	0
Sorenson	1.00	1.00	0	1.00	0	0
Dry						
Simple Matching	1.00	0.89	0	0.89	0.10	0.11
Sorenson	1.00	0.94	0	0.94	0	0

A = *Leptobrachium abotti* B = *Kalophrynus pleurostigma* C = *Bufo biporcatus*
D = *Rana limnocharis*

Table 6.15. Similarities between four dominant species based on the proportion of anurans abundance in the Closed Canopy Logged Plots between seasons at Camp 92, Central Kalimantan

Coefficient/Species similarity	AB	AC	AD	BC	BD	CD
Wet1						
Simple Matching	1.00	1.00	0	1.00	0	0
Sorenson	1.00	1.00	0	1.00	0	0
Wet2						
Simple Matching	1.00	1.00	0	1.00	0	0
Sorenson	1.00	1.00	0	1.00	0	0
Dry						
Simple Matching	1.00	1.00	0	1.00	0	0
Sorenson	1.00	1.00	0	1.00	0	0

Table 6.14 and 6.15 show that mutual species occurrence AB, AC and BC have a very high similarity because all these species were forest litter dependent anurans and occur together in the observed plots. However, species pairs AD, BD and CD have a great dissimilarity because *Rana limnocharis* (species D) did not occur in the Unlogged and Closed Canopy Logged Plots.

Table 6.16. Similarities between four dominant species based on the proportion of anurans abundance in the Open Canopy Logged Plots between seasons at Camp 92, Central Kalimantan

Coefficient/Species similarity	AB	AC	AD	BC	BD	CD
Wet1						
Simple Matching	0	0	0	1.00	1.00	1.00
Sorenson	0	0	0	1.00	1.00	1.00
Wet2						
Simple Matching	0	0	0	1.00	1.00	1.00
Sorenson	0	0	0	1.00	1.00	1.00
Dry						
Simple Matching	0	0	0	1.00	1.00	1.00
Sorenson	0	0	0	1.00	1.00	1.00

Table 6.16 indicates that mutual species occurrence BC, BD and CD have a very high similarity because all these species occur together in the observed plots. However, species pairs AB, AC and AD have a low similarity because *Leptobrachium abotti* (species A) did not occur in the Open Canopy Logged Plots.

6.7. Discussion

Abundance of terrestrial anurans

The pitfall trap results show that there was a reduction in abundance of forest-dependent anurans in the open canopy logged plots compared to the unlogged and closed canopy logged plots. These observed differences in abundance would likely be more pronounced in the dry season, when rainfall and humidity become patchy. Low terrestrial anuran abundances in the open canopy logged plots (Figure 6.3) may reflect unfavourable environmental conditions, possibly related to moisture stress (Figure 6.6). Anurans have a low tolerance for hot and dry conditions. Low humidity and high air temperature in the open canopy logged plots probably cause dehydration (Spotila 1972 and Petranka *et al.* 1993), which means some areas could be unsuitable for amphibians (Jaeger 1971, Feder 1983 and Welsh 1990). These factors limited the presence and movement of anurans, and confirmed the important role of moisture in anuran activity. The variation in abundance of anurans (Figures

6.1, 6.2 and 6.3) between seasons may reflect the variability in moisture regimes and numbers were lowest in open canopy logged plots in the dry season.

Riparian species are bathed constantly by water, but terrestrial amphibians have had to evolve adaptations to cope with the inevitable loss of body water while maintaining a moist skin for gas exchange. They are sensitive to water loss and often limited to areas where there is sufficient moisture for their survival and reproduction. In general, amphibians - especially the more terrestrial species - are found in patches of suitable habitat surrounded by conditions that are relatively harsh to them. Therefore, rainfall and humidity will significantly influence their activity and distribution patterns. Furthermore, because amphibians are ectotherms, they spend a great deal of time undergoing behavioral thermoregulation to avoid extremely high or extremely low temperatures (review in Duellman and Trueb 1994; Sinch 1990). The demands of water balance and thermoregulation will limit the movements of amphibians.

Species composition -The highest proportion of anuran species captured by pitfall trapping in the Unlogged Plots was represented by pelobatids with *Leptobrachium abotti* as the dominant species but in the Closed Canopy Logged Plots, *Kalophrynus pleurostigma* was the most dominant species. However, the biggest proportion of anuran species captured in the Open Canopy Logged Plots was represented by the bufonid *Bufo biporcatus*. Certain species appear to be characteristic of particular forest types. For example, The primary forest-dependent anuran *Leptobrachium abotti* was completely absent in the Open Canopy Logged Plots, whilst *Rana limnocharis* and *Pedostibes hosei* as pioneer species that were mainly found in disturbed habitat, was almost absent in the primary forest habitat of Unlogged Plots. *Bufo biporcatus* and *Kalophrynus pleurostigma* as wide-ranging species with broad range of thermal tolerance are more tolerant than frogs and more adaptable to the extreme conditions (Duellman and Trueb 1994), therefore both these species occurred in the Unlogged and Logged Plots.

Pitfall traps capture some species more easily than others (Karns 1986, Corn and Bury 1990, Dodd 1991). Anurans that are strong jumpers or climbers (e.g. *Gasthrophyne*, most *Rana*, most *Hyla*) are more difficult to trap than the more terrestrial species (e.g. *Bufo*, *Kalophrynus*, *Leptobrachium*) which lack these abilities (Franz and Ashton 1989, Dodd 1991). Therefore, it is likely that only a small portion of the species of ranid were captured in the pitfalls in the unlogged and logged plots. In addition, most of the ranids captured were found in the pitfalls primarily located near the streambanks, which suggests that they are the species that live primarily in this habitat.

Microhabitat associations-The strongest associations were found between litter cover and abundance of anurans. Statistical analysis showed that there was a very highly significant effect of litter cover on abundance of anurans. It is also interesting that the relationship was stronger during the dry season compared to the wet season. This suggests that the role of forest litter cover during dry periods is much more important than in the wet season for maintaining the humidity of the soil which is particularly important for anurans. Most forest anurans have poor dispersal abilities (relative to birds and mammals), so forest habitat use by the forest dependent anurans may be influenced heavily either by humidity or proximity to water.

Microclimatic associations-The strongest associations were found between air temperature and humidity. Statistical analysis also showed that there was a significant difference in abundance of anurans between plots during the dry season. The humidity was much lower but temperature higher during the dry season than in the wet season. The regression analysis indicated that the number of anurans captured was related to temperature and humidity. It also showed that the relationship was stronger in the dry season than in the wet season. The mixture of temperature and humidity affects the distribution and activity patterns of the anurans. Therefore, the capture rates of anurans during the dry season were much lower than in the wet season, so affecting abundance estimates.

VII. SEARCHED QUADRATS

7.1. Introduction

Quadrats were searched on the forest floor in the Unlogged and Logged Plots of Kayu Mas Forestry Logging Concession at Camp 92, Central Kalimantan, Indonesia as described in Chapter 4. Complete descriptions of the procedures adapted for use at the study sites are given by Inger and Colwell (1977) and Heyer *et al.* (1994).

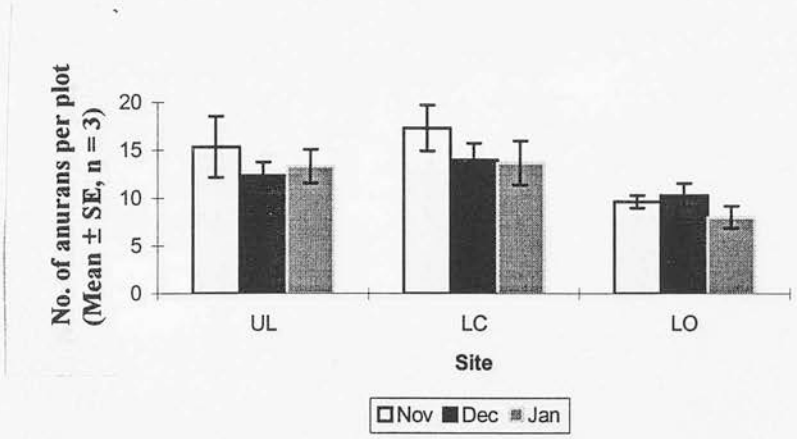
In this chapter, the results of sampling anurans by this method are presented in several sections ie. abundance of anurans, species richness and composition, and presence-absence data. Microclimate and vegetation components in relation to abundance of anurans and species richness are also discussed. Quadrat sampling yielded information essentially only on ground dwelling species and the very few arboreal anurans caught in quadrats are ignored in this analysis.

7.2. Abundance of anurans

7.2.1. *Difference in abundances*

The average number of anurans recorded in searched quadrats per study plot in the unlogged plots was similar to that in the closed canopy logged plots and only varied slightly between sites and between seasons as shown in Figures 7.1, 7.2 and 7.3. There were no significant differences on abundace of anurans between the unlogged and the closed canopy plots (see significance test Table 7.1).

Figure 7.1. Number of individual anurans (searched quadrat) in different sites during wet season 1 at Camp 92, Central Kalimantan



Bars are standard errors UL = Unlogged Plot LC = Closed Canopy Logged Plots LO = Open Canopy Logged Plot

Figure 7.2. Number of individual anurans (searched quadrat) in different sites during wet season 2 at Camp 92, Central Kalimantan

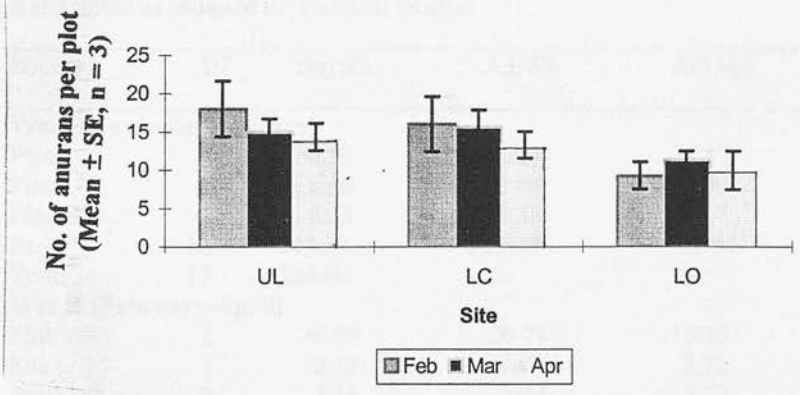
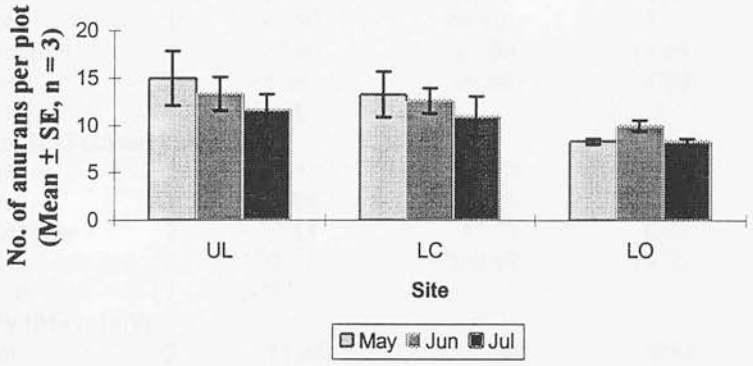


Figure 7.3 Number of individual anurans (searched quadrat) in different sites during the dry season at Central Kalimantan



There were no significant differences (GLM ANOVA, $P > 0.05$) in the abundance of anurans between Unlogged and Closed Canopy Logged Plots in different seasons as indicated in Table 7.1.

Table 7.1. Analysis of Variance (General Linear Model), significance test for abundance of anurans between Unlogged and Closed Canopy Logged sites in different seasons at Camp 92, Central Kalimantan as sampled by searched quadrat

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Wet I (November-January)						
Plot	2	64.33	64.33	32.17	2.69	0.108
Site	1	8.00	8.00	8.00	0.67	0.429
Plot*Site	2	8.33	8.33	4.17	0.35	0.712
Error	12	143.33	143.33	11.94		
Total	17	224.00				
Wet II (February-April)						
Plot	2	30.78	30.78	15.39	0.78	0.481
Site	1	2.72	2.72	2.72	0.14	0.717
Plot*Site	2	7.44	7.44	3.72	0.19	0.831
Error	12	237.33	237.33	19.78		
Total	17	278.28				
Dry (May-July)						
Plot	2	30.33	30.33	15.17	1.43	0.277
Site	1	4.50	4.50	4.50	0.42	0.527
Plot*Site	2	24.33	24.33	12.77	1.15	0.350
Error	12	127.33	127.33	10.61		
Total	17	186.50				

Table 7.2. Analysis of Variance (General Linear Model), significance test for abundance of anurans between Unlogged and Open Canopy Logged sites in different seasons at Camp 92, Central Kalimantan as sampled by searched quadrat

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Wet I (November-January)						
Plot	2	12.33	12.33	6.16	0.78	0.480
Site	1	84.50	84.50	84.50	10.71	0.007**
Plot*Site	2	27.00	27.00	13.50	1.71	0.222
Error	12	94.66	94.66	7.89		
Total	17	218.50				
Wet II (February-April)						
Plot	2	4.33	4.33	2.17	0.13	0.875
Site	1	138.89	138.89	138.89	8.65	0.012**
Plot*Site	2	12.11	12.11	6.06	0.38	0.694
Error	12	192.67	192.67	16.06		
Total	17	348.00				
Dry (May-July)						
Plot	2	17.44	17.44	8.72	1.50	0.263
Site	1	88.89	88.89	88.89	15.24	0.002**
Plot*Site	2	23.44	23.44	11.72	2.01	0.177
Error	12	70.00	70.00	5.83		
Total	17	199.78				

Table 7.3. Analysis of Variance (General Linear Model), significance test for abundance of anurans between Closed Canopy and Open Canopy Logged sites in different seasons at Camp 92, Central Kalimantan as sampled by searched quadrat

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Wet I (November-January)						
Plot	2	22.33	22.33	11.16	1.41	0.283
Site	1	144.50	144.50	144.50	18.19	0.001**
Plot*Site	2	20.33	20.33	10.16	1.28	0.313
Error	12	95.33	95.33	7.94		
Total	17	282.50				
Wet II (February-April)						
Plot	2	21.78	21.78	10.89	0.79	0.477
Site	1	102.72	102.72	102.72	7.43	0.018**
Plot*Site	2	21.78	21.78	10.89	0.79	0.477
Error	12	166.00	166.00	13.83		
Total	17	312.28				
Dry (May-July)						
Plot	2	6.78	6.78	3.39	0.54	0.594
Site	1	53.39	53.39	53.39	8.58	0.013**
Plot*Site	2	7.44	7.44	3.72	0.60	0.565
Error	12	166.66	166.66	9.26		
Total	17	142.28				

In contrast, the mean numbers of individual anurans captured per site in the Unlogged and the Closed Canopy Logged Plots were significantly different (GLM ANOVA, $P < 0.01$, $n = 27$) compared to the Open Canopy areas of Logged Plots (Table 7.2 and 7.3).

7.2.2. *Microclimate and abundance of anurans*

Humidity and temperature had significant effects on the abundance of anurans (Table 7.4 and 7.5).

Table 7.4. Analysis of Covariance for abundance of anurans in relation to humidity between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	Adj MS	F	P
Wet Season1					
Humidity	3	0.251	0.083	83.38	0.000***
Site	2	0.022	0.011	11.05	0.001***
Plot	2	0.0004	0.0002	0.02	0.981
Site*Plot	4	0.008	0.002	2.15	0.125
Error	15	0.015	0.001		
Total	26	0.643			
Wet Season2					
Humidity	3	0.407	0.135	38.63	0.000***
Site	2	0.032	0.016	4.59	0.028**
Plot	2	0.0003	0.0001	0.05	0.952
Site*Plot	4	0.0019	0.0004	0.14	0.966
Error	15	0.052	0.003		
Total	26	0.794			
Dry Season					
Humidity	3	0.194	0.064	42.28	0.000***
Site	2	0.032	0.016	10.50	0.001***
Plot	2	0.0002	0.0001	0.07	0.936
Site*Plot	4	0.015	0.0038	2.49	0.088
Error	15	0.023	0.0015		
Total	26	0.462			

Table 7.5. Analysis of Covariance for abundance of anurans in relation to temperature between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	Adj MS	F	P
Wet Season 1					
Temperature	3	0.259	0.086	181.21	0.000***
Site	2	0.016	0.008	17.66	0.000***
Plot	2	0.0005	0.0002	0.05	0.950
Site*Plot	4	0.0005	0.0001	0.30	0.873
Error	15	0.007	0.004		
Total	26	0.643			
Wet Season 2					
Temperature	3	0.435	0.145	89.79	0.000***
Site	2	0.036	0.018	11.31	0.001***
Plot	2	0.001	0.0005	0.42	0.665
Site*Plot	4	0.002	0.0005	0.43	0.787
Error	15	0.024	0.0016		
Total	26	0.794			
Dry Season					
Temperature	3	0.197	0.065	49.60	0.000**
Site	2	0.012	0.006	4.86	0.024**
Plot	2	0.003	0.001	1.24	0.318
Site*Plot	4	0.007	0.0018	1.40	0.280
Error	15	0.019	0.0013		
Total	26	0.462			

The combination of humidity and temperature determines the rate of water loss from an amphibian's surface especially for the forest-dependent anurans. For this reason, the amount of moisture in the air strongly affects distribution and activity patterns (Heyer *et al.* 1994). Soil temperature is also critical for the terrestrial anurans because it significantly influences anuran distribution and activity patterns. Therefore, the role of humidity and temperature is very important in affecting the abundance of anurans.

Regression analysis was used to examine the relationship between humidity and the abundance of anurans during the dry and wet seasons (Figures 7.4, 7.5 and 7.6).

Fig. 7.4. Regression analysis for humidity and abundance of anurans during wet season 1 at Camp 92, Central Kalimantan

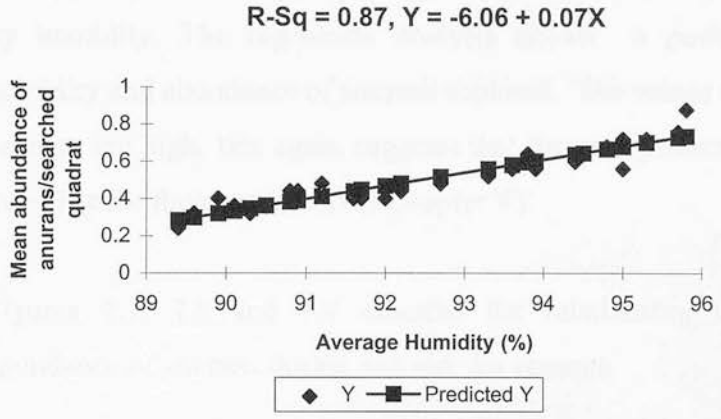


Fig. 7.5. Regression analysis for humidity and abundance of anurans during wet season 2 at Camp 92, Central Kalimantan

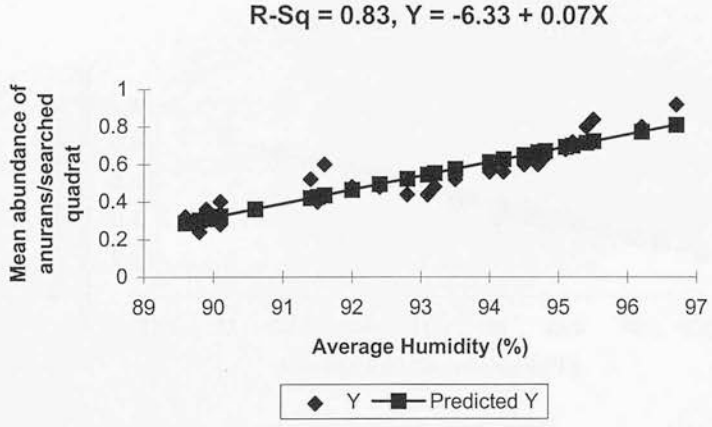
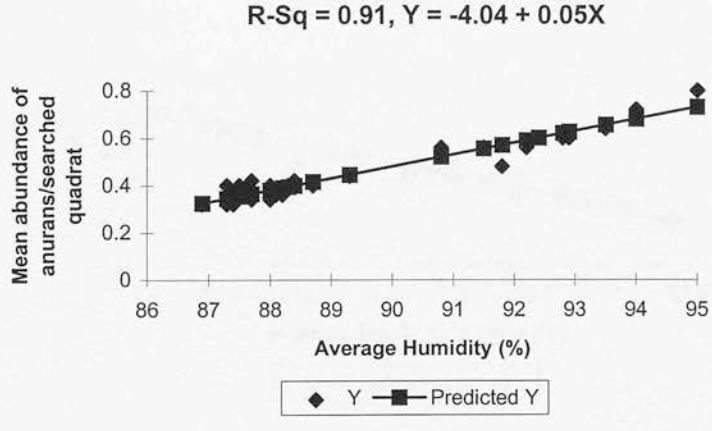


Fig. 7.6. Regression analysis for humidity and abundance of anurans during the dry season at Camp 92, Central Kalimantan



The statistical evidence (R-sq = 0.87, 0.83 and 0.91, p-value < 0.0001, n = 27) strongly supports the hypothesis that the number of anurans captured was influenced by humidity. The regression analysis shows a positive relationship between humidity and abundance of anurans captured. The values of R-Sq in the wet and dry seasons are high, this again suggests that forest-dependent anurans depend on high humidity for their survival (see Chapter V).

Figures 7.7, 7.8 and 7.9 describe the relationship between temperature and abundance of anurans during wet and dry seasons.

Fig. 7.7. Regression analysis for temperature and abundance of anurans during wet season1 at Camp 92, Central Kalimantan

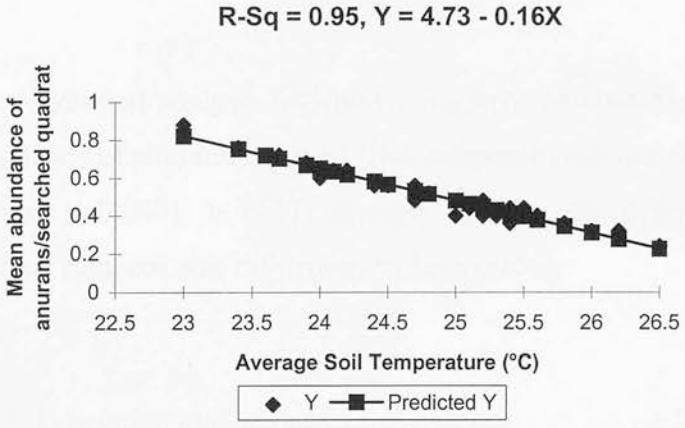


Fig. 7.8. Regression analysis for temperature and abundance of anurans during wet season2 at Camp 92, Central Kalimantan

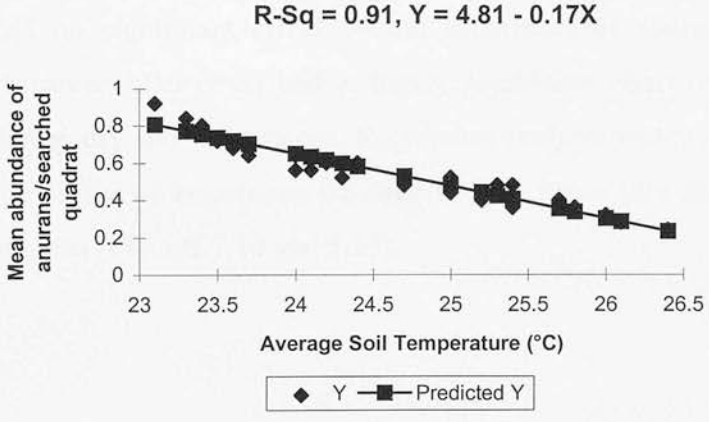
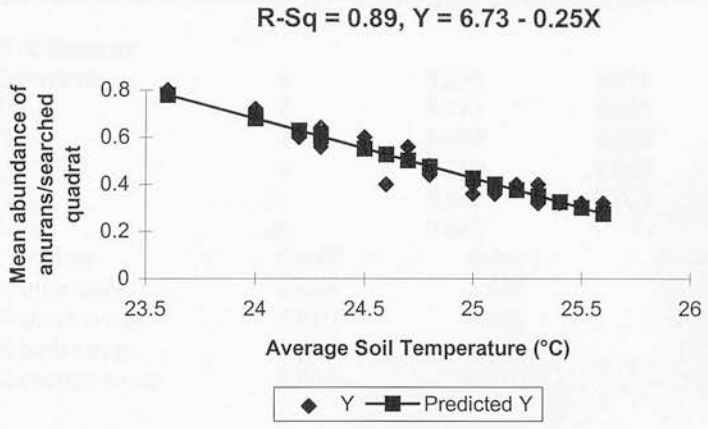


Fig. 7.9. Regression analysis for temperature and abundance of anurans during the dry season at Camp 92, Central Kalimantan



The regression analysis indicates a negative relationship between temperature and abundance of anurans captured. The statistical evidence (R-sq = 0.95, 0.91 and 0.89, p-value < 0.0001, n = 27) strongly supports the hypothesis that the number of anurans captured was influenced by temperature.

7.2.3. *Vegetation and abundance of anurans*

Table 7.6 indicates that % plant cover (shrub cover, herb cover and canopy cover) had no significant effect on the abundance of anurans (ANCOVA, P > 0.05). However, litter cover had a highly significant effect on the abundance of anurans during dry and wet seasons. Regression analysis was undertaken in order to indicate the degree of association between % litter cover (dry season) and the abundance of anurans (Figures 7.10 and 7.11).

Table 7.6. Analysis of Covariance for abundance of anurans in relation to % plant cover between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet Seasons					
Covariates	4	0.236	0.059	27.29	0.000***
Site	2	0.015	0.007	3.49	0.059
Plot	2	0.009	0.004	2.17	0.151
Site*Plot	4	0.009	0.002	1.12	0.384
Error	14	0.030	0.002		
Total	26	0.643			
Covariate	Coeff	Stdev	t-value		P
% litter cover	0.029	0.039	10.155		0.000***
% shrub cover	0.010	0.006	1.732		0.105
% herb cover	0.006	0.005	1.191		0.253
% canopy cover	0.006	0.010	0.664		0.517
Dry Season					
Covariates	4	0.185	0.046	20.10	0.000***
Site	2	0.023	0.011	5.09	0.022**
Plot	2	0.006	0.003	1.45	0.268
Site*Plot	4	0.007	0.001	0.86	0.511
Error	14	0.032	0.002		
Total	26	0.462			
Covariate	Coeff	Stdev	t-value		P
% litter cover	0.031	0.004	7.499		0.000***
% shrub cover	0.001	0.005	0.191		0.851
% herb cover	0.012	0.005	2.241		0.052
% canopy cover	0.003	0.007	0.141		0.688

Fig. 7.10. Regression analysis for % litter cover and abundance of anurans during wet seasons at Camp 92, Central Kalimantan

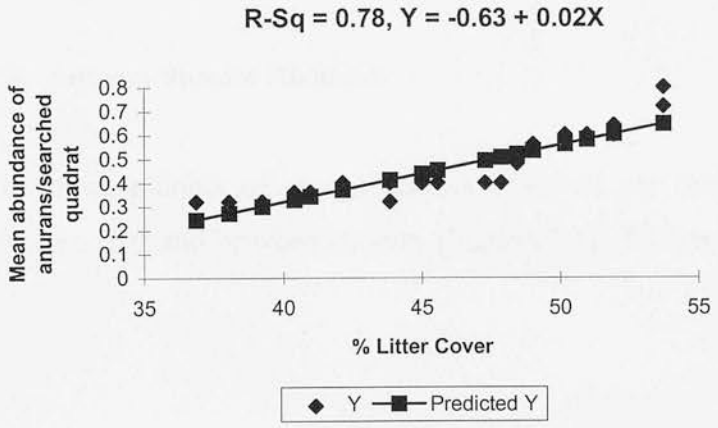
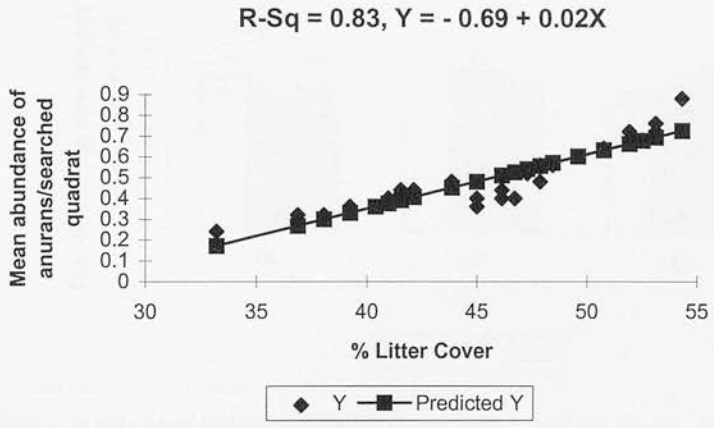


Fig. 7.11. Regression analysis % litter cover and abundance of anurans during the dry season at Camp 92, Central Kalimantan

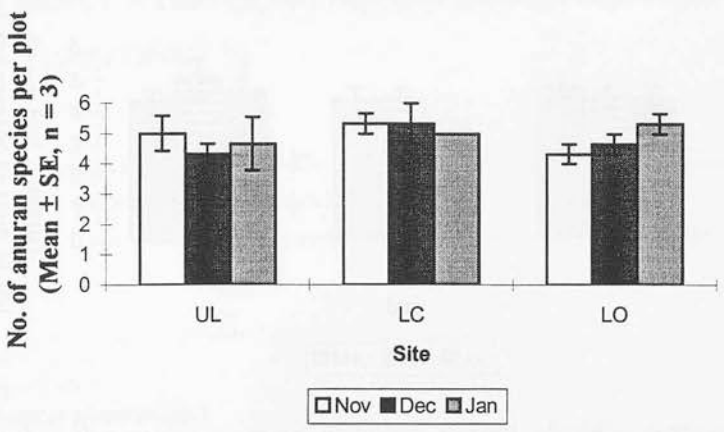


The regression shows a positive relationship in which increased % litter cover leads to an increased number of anurans captured The regression analysis at R-sq = 0.78 (wet season) and 0.83 (dry season)(p-value < 0.0001, n = 27) also supports the hypothesis that the number of anurans captured was associated with % litter cover. It is interesting that this relationship is stronger in the dry season than in the wet season. This suggests that the function of litter cover during the dry season is more important than in the wet season, and may be related to its ability to maintain the humidity which is important for anurans.

7.3. Anuran Species Richness

The mean number of anuran species observed per searched plot varied slightly between sites and between seasons (Figures 7.12, 7.13, and 7.14)

Figure 7.12. Number of anuran species (searched quadrat) in different sites during wet season 1 at Camp 92, Central Kalimantan



Bars are standard errors UL = Unlogged Plot LC = Closed Canopy Logged Plot LO = Open Canopy Logged Plot

Figure 7.13. Number of anuran species (searched quadrat) in different sites during wet season 2 at Camp 92, Central Kalimantan

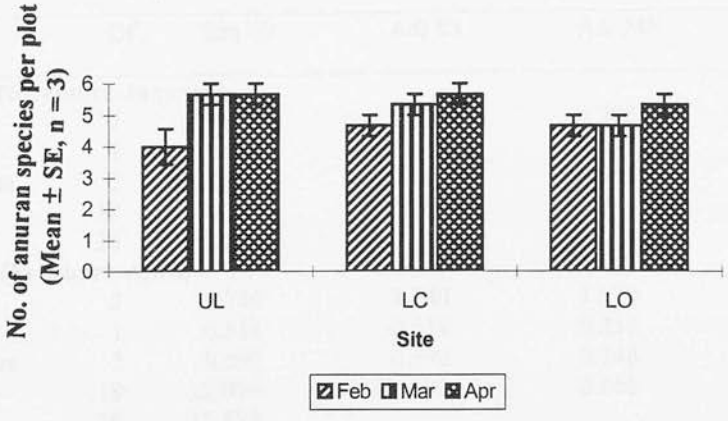
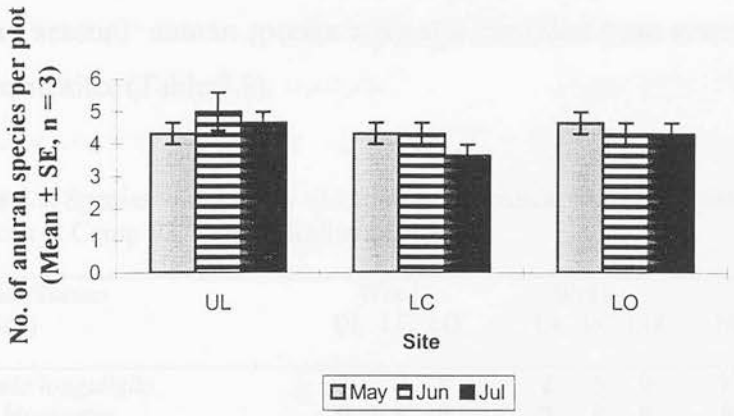


Figure 7.14. Number of anuran species (searched quadrat) in different sites during the dry season 2 at Camp 92, Central Kalimantan, Indonesia



Although the mean species richness varied slightly between sites in different seasons, there was no significant difference (GLM ANOVA, $P > 0.05$,) either between sites or between seasons as indicated in Table 7.7. Thus, statistical analysis indicates that species richness did not differ significantly among sites and no significant seasonal effects on anuran species richness were observed.

Table 7.7. Analysis of Variance (General Linear Model), significance test for species richness among sites in different seasons at Camp 92, Central Kalimantan

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Wet I (November-January)						
Plot	2	0.222	0.222	0.778	1.11	0.353
Site	1	1.555	1.555	2.722	2.72	0.125
Plot*Site	2	2.222	2.222	0.555	0.79	0.547
Error	18	12.666	12.666	0.703		
Total	26	16.666				
Wet II (February-April)						
Plot	2	2.740	2.740	1.370	2.06	0.157
Site	1	0.518	0.518	0.259	0.39	0.683
Plot*Site	2	0.592	0.592	0.148	0.22	0.922
Error	18	12.000	12 .000	0.666		
Total	26	15.852				
Dry (May-July)						
Plot	2	0.074	0.074	0.037	0.08	0.920
Site	1	0.518	0.518	0.259	0.58	0.568
Plot*Site	2	1.037	1.037	0.259	0.58	0.679
Error	18	8.000	8.000	0.444		
Total	26	9.629				

Totals of 10, 8, and 9 species of anurans were observed in searched quadrats during Wet Season 1 in the Unlogged Plots, Closed canopy Logged Plots, and Open canopy Logged Plots, respectively . Totals of 7, 7, and 9 (wet season 2) and 8, 7 and 7 (dry season) anuran species were also recorded from searched quadrats in three different sites (Table 7.8).

Table 7.8. Species and number of captures per season from three sites in different seasons at Camp 92, Central Kalimantan

Species/Season (Site)	Wet 1			Wet 2			Dry		
	UL	LC	LO	UL	LC	LO	UL	LC	LO
Ansonia longidigita	3	4	8	2	5	9	3	3	-
Bufo biporcatus	9	9	9	9	9	9	9	9	9
Kalophrynus pleurostigma	9	9	9	9	9	9	9	9	9
Leptobrachium abotti	9	9	-	9	9	-	9	9	-
Megophrys nasuta	3	7	1	8	5	1	4	3	-
Pedostibes rugosus	-	-	9	-	-	9	-	-	9
R. blythi	2	-	-	-	-	-	-	-	-
Rana finchi	2	2	-	-	-	-	2	3	-
R. chalconota	7	6	4	6	8	5	1	1	2
R. palavanensis	2	3	-	3	2	-	2	-	-
R. limnocharis	1	-	8	-	-	9	-	-	8
R. kuhli	-	-	4	-	-	1	-	-	1
R. signata	-	-	-	-	-	-	-	-	2
Staurois natator	-	-	1	-	-	2	-	-	-
Total species	10	8	9	7	7	9	8	7	7

A total of only 14 species was recorded in the searched quadrats (Table 7.8), probably representing only a small portion of the total Bornean anurans in the study plots. Of these, eight species (*Rana blythi, R. chalconota, R. finchi, R. kuhli, R. limnocharis, R. palavanensis, R. signata and Staurois natator*, family *Ranidae*) were ranids, three were bufonids (*Bufo biporcatus*, *Ansonia longidigita*, and *Pedostibes rugosus*, family Bufonidae), two were pelobatids (*Leptobrachium abotti* and *Megophrys nasuta*, family *Pelobatidae*) and only one species was a microhylid (*Kalophrynus pleurostigma*, family *Microhylidae*).

The proportion of forest-dwelling anurans grouped according to their families, recorded from searched quadrats is presented in Figures 7.15, 7.16 and 7.17. Of the

total number of species recorded from searched quadrats in the unlogged plots, 35.5% of them were represented by pelobatids, 26.4% microhylids, 28.0% bufonids and the smallest proportion belonged to the ranids. In the closed canopy areas of logged plots, 30.4% were pelobatids, 33.5% microhylids, 27.1 bufonids and the rest belonged to the ranids. In contrast, the largest proportion in the open canopy logged plots were ranids (43.9%, of which 40% was *Rana limnocharis*) and bufonids (42.3%, of which around 39% was dominated by *Bufo biporcatus*). Only 13.7% of anurans recorded there were microhylids and the smallest proportion belonged to the pelobatids.

Fig 7.15 The proportion of individual anurans in each family recorded from searched quadrats in different sites during wet season I at Camp 92, Central Borneo

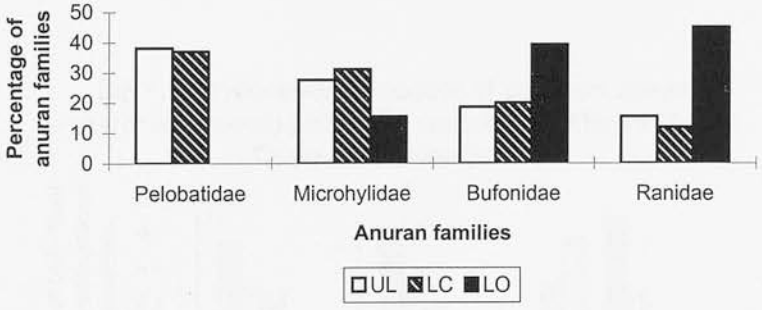


Fig 7.16 The proportion of individual anurans in each family recorded from searched quadrats in different sites during wet season 2 at Camp 92, Central Borneo

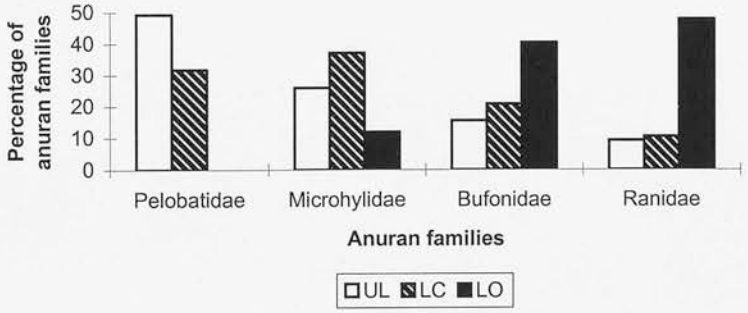
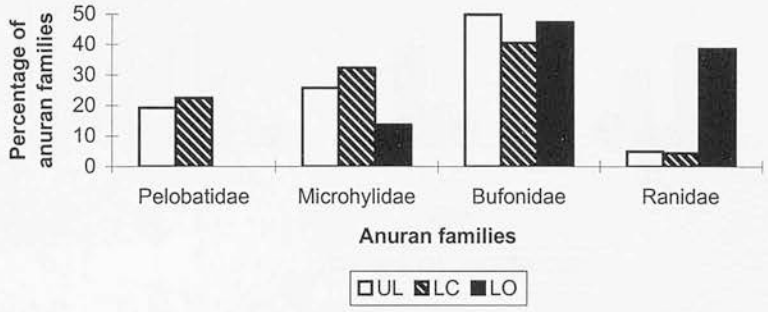
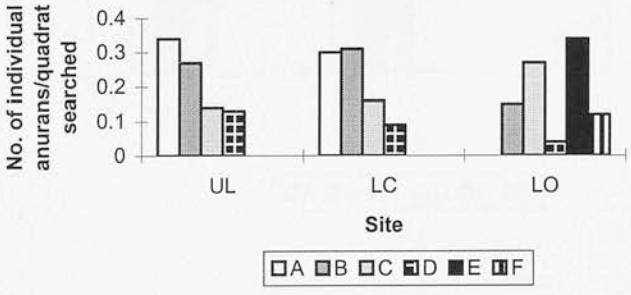


Fig 7.17 The proportion of individual anurans in each family recorded from searched quadrats in different sites during the dry season at Camp 92, Central Borneo



Figures 7.18, 7.19 and 7.20 show species composition between seasons for the dominant species in terms of relative density.

Fig 7.18 Frequency distribution of dominant species (searched quadrat) during wet season 1 in different sites at Camp 92, Central Borneo



A = *Leptobrachium abotti* B = *Kalophrynus pleurostigma* C = *Bufo biporcatus* D = *Rana chalconota* E = *Rana limnocharis* F = *Pedostibes rugosus*

Fig 7.19 Frequency distribution of dominant species (searched quadrat) during wet season 2 at Camp 92, Central Borneo

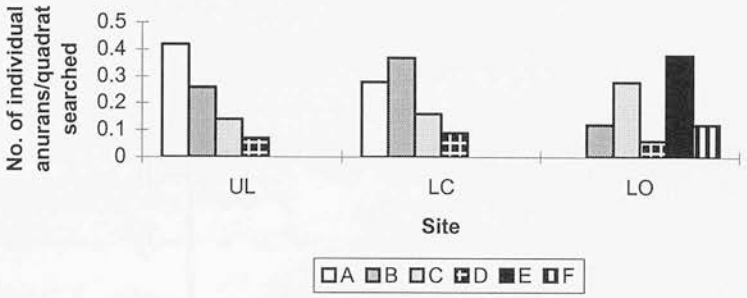
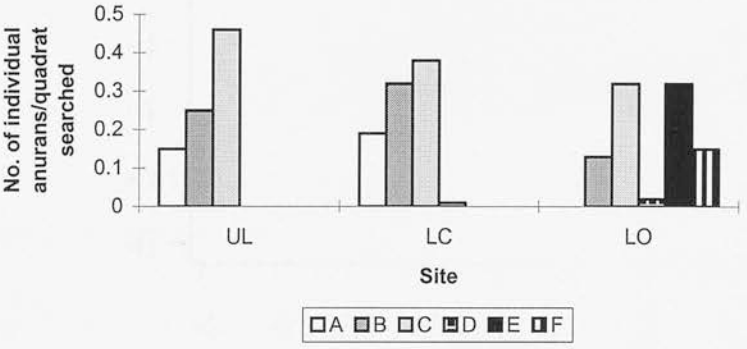
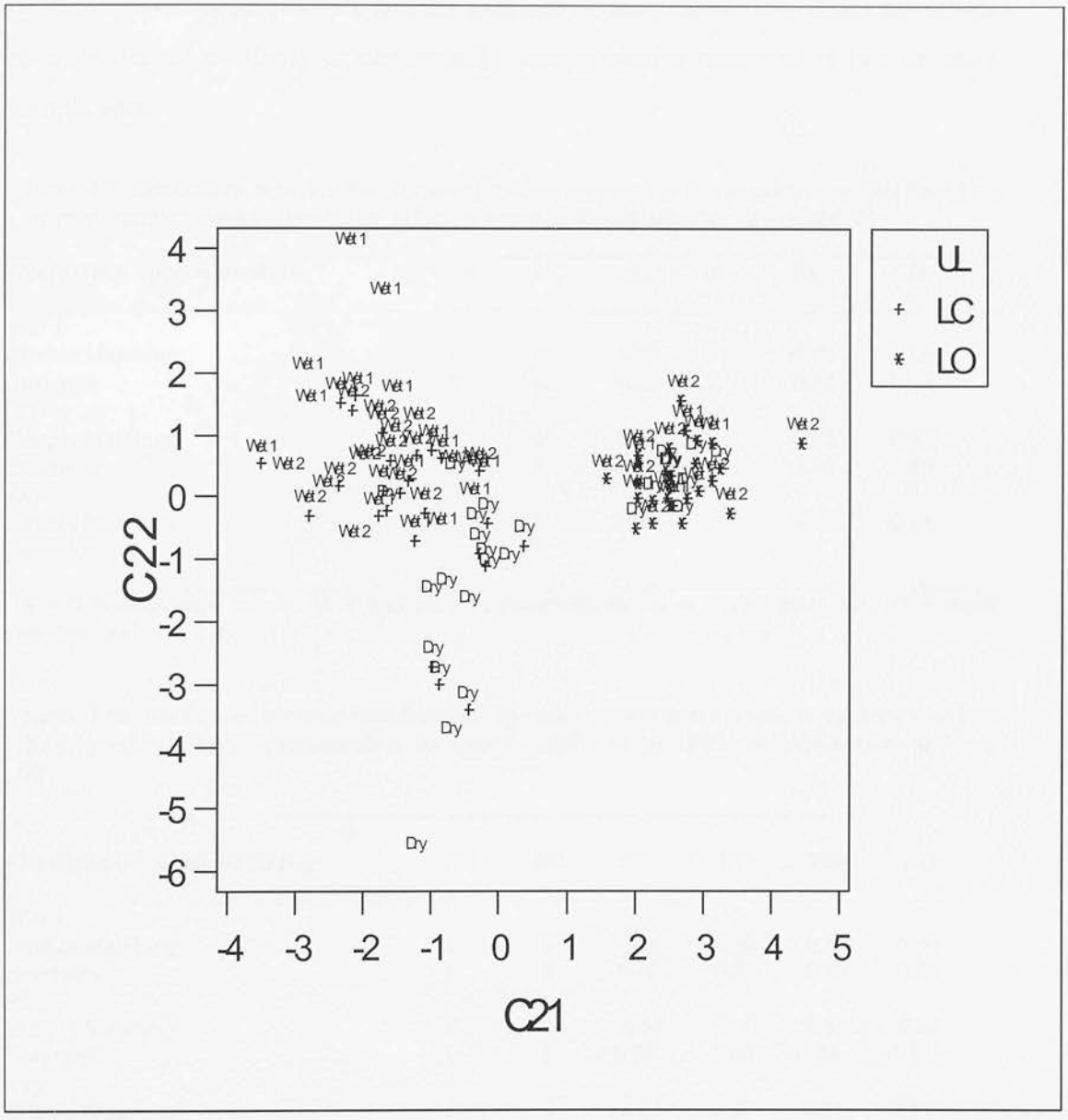


Fig 7.20 Frequency distribution of dominant species (searched quadrat) during the dry season at Camp 92, Central Borneo



From the figures above it can be seen that the dominant species observed (by searched quadrats) during the wet season in the Unlogged and Closed Canopy Logged Plots were *Leptobrachium abotti* and *Kalophrynus pleurostigma*. However, during the dry season, *Bufo biporcatus* was observed to be more dominant in the Unlogged and Logged sites. *Rana limnocharis* was also the dominant species observed in the searched quadrat of Open Canopy Logged Plots.

Fig 7.21 PCA Chart for presence-absence of species (Searched quadrats) between seasons in different sites at Camp 92, Central Kalimantan



C21 = PCA Axis 1 (Site) C22 = PCA Axis 2 (Season)

Figure 7.21 shows that the dominant species (*Leptobrachium abotti*) present in the unlogged plots did not mix with the dominant species (*Rana limnocharis*) found in the logged plots. The distribution of species in the Open Canopy Logged Plots seemed to be more concentrated while in the Unlogged and Closed Canopy Logged Plots they tended to be distributed sparsely. The distribution patterns seemed to be related to the shading and moisture conditions.

Similarity coefficients used to show the coincidence of occurance between species pairs in the searched quadrats and the measure of association expressed the faunal resemblance as similarity or dissimilarity among species observed at two or more sample sites.

Table 7.9. Similarities between four dominant species recorded in the searched quadrats based on the proportion of anurans abundance in the unlogged plots between seasons at Camp 92

Coefficient/ Species similarity	AB	AC	AD	BC	BD	CD
Wet 1						
Simple Matching	1	1	0.77	1	0.77	0.88
Sorenson	1	1	0.87	1	0.87	0.94
Wet 2						
Simple Matching	1	1	0.66	1	0.66	0.66
Sorenson	1	1	0.80	1	0.80	0.80
Dry						
Simple Matching	1	1	0.11	1	0.11	0.11
Sorenson						

A = *Leptobrachium abotti* B = *Kalophrynus pleurostigma* C = *Bufo biporcatus* D = *Rana chalconota*

Table 7.10 Similarities between four dominant species recorded in the searched quadrats based on the proportion of anurans abundance in the Closed Canopy Logged Plots between seasons at Camp 92

Coefficient/ Species similarity	AB	AC	AD	BC	BD	CD
Wet 1						
Simple Matching	1	1	0.66	0.66	0.66	0.66
Sorenson	1	1	0.80	0.80	0.80	0.80
Wet 2						
Simple Matching	1	1	0.88	1.00	0.88	0.88
Sorenson	1	1	0.94	1.00	0.94	0.94
Dry						
Simple Matching	1	1	0.11	1.00	0.11	0.11
Sorenson	1	1	0.20	1.00	0.20	0.20

A = *Leptobrachium abotti* B = *Kalophrynus pleurostigma* C = *Bufo biporcatus* D = *Rana chalconota*

Table 7.9 and 7.10 show that coincidence of occurrence between species pairs AB, AC, BC was very high similar for any season because all these species occur together at ground level in the observed plots. However, species pairs AD, BD and CD during the wet season were not coincident. This might be because *Rana chalconota* (species D) is a semi-arboreal species whilst species A, B and C are

forest litter-dependent anurans. In addition, the level of coincidence of species pairs AD, BD and CD was very low during the dry season. This may be because all the species were sparsely distributed because of unfavourable drying conditions.

Table 7.11 Similarities between the four dominant species recorded in the searched quadrats based on the proportion of anuran abundance in the Open Canopy Logged Plots between seasons at Camp 92, Central Kalimantan

Coefficient/ Species similarity	AB	AC	AD	BC	BD	CD
Wet 1						
Simple Matching	0	0	0	1	0.44	0.44
Sorenson	0	0	0	1	0.61	0.61
Wet 2						
Simple Matching	0	0	0	1	0.55	0.55
Sorenson	0	0	0	1	0.71	0.71
Dry						
Simple Matching	0	0	0	1	0.22	0.22
Sorenson	0	0	0	1	0.36	0.36

A = *Leptobrachium abotti* B = *Kalophrynus pleurostigma* C = *Bufo biporcatus* D = *Rana chalconota*

Table 7.11 indicates that the coefficient of similarity for species AB, AC, AD was zero because *Leptobrachium abotti* (Species A) was absent in the observed plots. In contrast, coefficients for species B and C were very high because they occurred together in the same habitat (forest litter).

7.3.2. Microclimate and Species Richness

Humidity and temperature had no significant effects (ANCOVA, $P > 0.05$) on species richness as indicated in Tables 7.12 and 7.13.

Table 7.12. Analysis of Covariance for species richness in relation to humidity between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet Season 1					
Humidity	3	3.35	1.11	1.80	0.191
Site	2	1.73	0.86	1.39	0.279
Plot	2	0.79	0.39	0.64	0.541
Site*Plot	4	1.01	0.25	0.41	0.800
Error	15	9.31	0.62		
Total	26	16.66			
Wet Season 2					
Humidity	3	1.83	0.61	0.90	0.463
Plot	2	3.07	1.53	2.27	0.138
Site	2	0.37	0.18	0.27	0.765
Plot*Site	4	3.92	0.98	1.45	0.267
Error	15	10.16	0.67		
Total	26	18.66			

Table ....contd.

Source	Df	Adj SS	MS	F	P
Dry Season					
Humidity	3	1.35	0.45	0.85	0.488
Plot	2	1.43	0.71	1.35	0.289
Site	2	0.08	0.04	0.08	0.923
Plot*Site	4	2.62	0.65	1.23	0.338
Error	15	7.97	0.53		
Total	26	12.51			

Table 7.13. Analysis of Covariance for species richness in relation to temperature between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet Season 1					
Temperature	3	2.05	0.68	0.97	0.434
Site	2	2.19	1.09	1.55	0.244
Plot	2	0.61	0.30	0.44	0.655
Site*Plot	4	1.38	0.34	0.49	0.743
Error	15	10.61	0.70		
Total	26	16.66			
Wet Season 2					
Temperature	3	0.30	0.10	0.13	0.939
Site	2	2.85	1.42	1.83	0.195
Plot	2	0.06	0.03	0.04	0.961
Site*Plot	4	3.30	0.82	1.06	0.409
Error	15	11.69	0.78		
Total	26	18.66			
Dry Season					
Temperature	3	1.49	0.49	0.95	0.440
Plot	2	0.87	0.44	0.84	0.451
Site	2	0.09	0.04	0.09	0.916
Plot*Site	4	2.06	0.51	0.99	0.445
Error	15	7.83	0.52		
Total	26	12.51			

Vegetation variables i.e. percentages of shrub cover, herb cover and canopy cover have no significant effect (ANCOVA, $P > 0.05$) on species richness but there was a significant effect of litter cover on species richness during the dry season as indicated in Table 7.14.

Table 7.14. Analysis of Covariance for species richness in relation to vegetation variables between seasons in different sites at Camp 92, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet Seasons					
Covariates	4	4.27	1.06	1.78	0.189
Site	2	0.64	0.32	0.54	0.594
Plot	2	0.51	0.25	0.43	0.662
Plot*Site	4	1.15	0.29	0.48	0.748
Error	14	8.39	0.59		
Total	26	16.66			
Covariate	Coeff	Stdev	t-value	P	
% litter cover	0.10	0.06	1.58	0.135	
% shrub cover	0.16	0.10	1.54	0.144	
% herb cover	0.10	0.09	1.14	0.271	
% canopy cover	0.23	0.16	1.36	0.195	
Dry Season					
Covariates	4	3.51	0.87	2.74	0.071
Site	2	0.73	0.36	1.14	0.348
Plot	2	1.05	0.52	1.65	0.228
Site*Plot	4	1.96	0.49	1.53	0.247
Error	14	4.48	0.32		
Total	26	9.62			
Covariate	Coeff	Stdev	t-value	P	
% litter cover	0.15	0.05	3.06	0.006**	
% shrub cover	0.11	0.07	1.56	0.141	
% herb cover	0.07	0.06	1.25	0.231	
% canopy cover	0.008	0.088	0.09	0.929	

7.4. Discussion

Although the survey technique used by active searched quarats was different from the passive sampling effort by pitfall traps, the results reported in this chapter strongly support those presented in chapter 6 .

The mean number of individual anurans observed per quadrat searched between the Unlogged and the Closed Canopy Logged Plots was not significantly different (Table 7.1), but they were significantly different compared to the Open Canopy Logged Plots (Table 7.2 and 7.3). The number of individual anurans per quadrat searched between seasons was much higher in the Unlogged and Closed Canopy

Logged Plots compared to the Open Canopy Logged Plots (Figures 7.1, 7.2 and 7.3). These results were similar with the results obtained in the pitfall traps (Chapter 6). This result is in line with the results of most studies examining the effects of timber harvesting on amphibians, which have reported a higher abundance in forest control plots than in harvested plots by using active sampling methods like area- and time-constrained searches (Corn and Bury 1989; Dodd 1991, Petranka et al. 1993, 1994 and Dupuis *et al.* 1995)(see Chapter I section 1.3). The difference in abundance of anurans was possibly due to the difference in humidity and temperature between sites as indicated in table 7.4 and 7.5. The difference was mainly due to the presence of large canopy gaps in the open canopy logged plots (see Chapter V). As a result, the humidity was much lower, but the temperature much higher, in the open canopy logged plots in comparison to the unlogged and closed canopy logged plots. The combination of humidity and temperature is likely to affect the abundance of anurans as explained in this Chapter section 7.2.2. In addition, Duellman and Trueb (1994) stated that the activity and distribution patterns of forest-dependent anurans are strongly influenced by the humidity and temperature of their surrounding environment.

Forest litter cover also had a significant effect on the abundance of anurans captured in the searched quadrat at any season (Table 7.6). Several activities of timber operations may affect the amphibian's forest floor habitat. Logging activity which creates large canopy gaps usually decreases plant litter, because intensive logging practices disrupted forest litter and reduced inputs from the leaves that are falling to the forest floor. This will certainly affect the activity of forest litter anurans that are very highly dependent on the presence of forest litter cover for their main refugia. In addition, drying of leaf litter and humus layers due to canopy removal limits amphibian movements and their ability to forage as well as reducing their abundance. Individuals not directly impacted by the immediate logging operation are probably subjected to the stresses associated with reduced or altered prey resources and changes in the physical characteristics of the soil and forest litter ecosystem. Therefore, the numbers of amphibians in such areas of open-canopy logged plots may be reduced. The results of this study is in line with the results from several

studies (Heatwole 1962; Pough *et al.*; 1987, De Graaf and Rudis 1990; Bonin 1991) that showed forest litter had important relationship with amphibian abundance. Their results suggest that intensive forest harvesting practices that minimize soil compaction and litter disruption might shorten the length of recovery time for amphibian species associated with this microhabitat. After intensive logging, for example, reduced inputs combined with increased rates of decomposition lead to a decline in forest litter. The time required for litter recovery to predisturbance levels can be quite long in some forest types, e.g., up to 50-80 years in northern hardwood forests (Likens et al. 1978; Federer 1984; Hughes and Fahey 1994). In addition, most of the forest anuran species captured were species dependent on water. These species may react unfavourably to the alteration of hydroperiods, increased insolation, and decreased relative humidity. Regression analysis (Figures 7.5 and 7.6) showed that there was a strong relationship between the percentage of litter cover and the abundance of anurans in the dry and wet seasons. This suggests that the function of litter is very important and possibly related to its ability to maintain humidity and it is also an important source of moisture throughout the season (Feder 1983; Corn and Bury 1990)

Statistical analysis showed that species richness did not differ among sites. This was possibly because selective logging does not reduce drastically the diversity of habitat in comparison to clear felling which in turn may affect species richness. The number of amphibian species in a certain habitat is mostly affected by the diversity of that habitat (Blaustein 1994). Since the effects of selective logging had not significantly influenced the diversity of the habitat, it is understandable that logging did not affect species richness. Although species richness did not differ in the treatment plots, species composition (see table 7.8) and the proportion of forest dwelling anurans Figures 7.15, 7.16 and 7.17) were clearly different. For example, the primary forest-dependent species (ie. *Leptobrachium abotti*) was not found in the open canopy logged plots (see also Figure 7.21) and in contrast, the disturbed forest species (ie. *Rana limnocharis*) was not found in the primary forest of unlogged plots. Apparently, opening of the canopy favors a few species to the detriment of the majority of forest-dependent species. Bury (1983) attributed the differences to

microclimatic changes, such as greater light penetration on the forest floor in the open canopy logged plots, leading to increased soil temperature and humidity. Such conditions could be unsuitable for *Leptobrachium abotti*, a species which Inger and Stuebing (1989) proposed should serve as a more sensitive indicator of vital limits and optimum climate than would most other anurans. In contrast, *Rana limnocharis* is a disturbed forest species occupying a wider range of habitat and has a higher operating temperature tolerance (Stebin and Cohen 1995). This is perhaps related to an ability of this frog to store and reabsorb large quantities of water in the bladder (up to 20-30% of its body mass: Zug 1993), which allows a greater tolerance of warmer and dryer microclimates in the open canopy logged plots.

CHAPTER VIII

NIGHT RIPARIAN TRANSECT

8.1. Introduction

The night riparian transect technique was applied to sampling anurans in streams flowing through the Unlogged Stands of Permanent Sample Plots at Camp 48 and the 2-year-old Selectively Logged Stands. Because streams in logged stands flowed through both closed and open-canopy areas, separate data were not available using this sampling method. Complete descriptions of the procedures of this riparian sampling were given in Chapter IV (Research Methods), and the technique and procedure used in this study followed the Sampling Methods for Amphibians in the Pacific Northwest (Bury and Corn 1991).

This chapter covers the results of night riparian sampling in several sections: physical characteristics and stream variables, abundance of anurans, species richness and composition, species presence-absence and distribution. In addition, this chapter describes the relationship between microclimate and vegetation components in relation to the abundance and species richness of anurans. Night riparian sampling yielded information only on adult anurans and the very few young anurans caught in the surveyed transect are ignored in this analysis.

8.2. Physical Characteristics of streams

8.2.1. *Streams in uncut forests (Permanent Sample Plots)*

Undisturbed headwater streams from the primary forest in the study site were characterized by continuous canopy cover, steep gradients, rocky substrates and a large amount of woody debris. The average width of the observational small streams (varied between 3-5m) sampled was about 4m and of the largest stream (varied

between 7-10m) was about 8m. The large streams had a permanent, moderate flow and ran through either alternating riffles (shallow water flowing rapidly and constantly over a rocky substrate) or pools (calm areas with clear water and relatively slow flow rates) of pristine tropical forest over bottoms of sand, gravel, boulders and bedrock. The small streams had impermanent water supplies and were largely dependent on rainfall events to fill the stream beds. The water rarely covered more than half of the beds when not in flood. The depth of water varied considerably with time because of frequent heavy rains. The water in these streams was usually clear except after rains. Precipitation sometimes exceeded 2.5cm on certain days between November 1994 and January 1995. Each rainfall event of this magnitude caused the water to rise rapidly, to become turbid and to increase sharply in velocity. When not in spate, and, hence, when the water was clear, the large streams were 0.5 - 1.0m deep in the deepest pools.

The bank on each observation stream varied from low and flat to high (in excess of 1.5m) and steep. Banks were generally clay, but at intervals were formed of rock outcrops. The forest arched over each stream shading around 80% of the bed. Within the surveyed zones, the widths of the beds were almost the same ranging from 4m to 8m. The beds of the small streams were comprised of sandy clay bottom and rocky substrates mainly pebbles (33-64mm) and gravel (5-16mm). While the large streams consisted of sandy soil and rocky substrate with gravel (17-32mm), pebbles, cobble (65-256mm) and head-sized boulder (>256mm).

8.2.2. *Streams in the 2-year-logged stands*

Physical comparisons between types of streams in unlogged and logged stands were similar, except that streams in logged stands had generally finer (silt) substrate, resulting from increased sedimentation with high silt deposit. The water of the streams flowing through the logged stands was more turbid. Some parts of the streams were blocked with fallen logs and other logging debris and covered with soil sediment. The depth of water varied with time because of the high intensity and frequency of rainfall events. Maximum pool depth was 1m, though most pools were

only 0.4 to 0.6m. The forest arched over each stream shading around 60% and the banks in general were exposed to the light.

The height of banks varied from 0.3m to 1m above the bed. The beds of the large streams comprised of sands and rocky substrates with mainly cobble and gravel with deep fine sediment on the bottom substrate; and the small stream consisted of rocky substrate in particular of pebbles and gravel with a high percentage of fine sediment. The stream bed width mainly varied between 3 and 5m.

8.3. Stream Variables

No significant differences existed in water temperature, depth (except for large stream) and width of the stream in the surveyed section between unlogged and logged over areas (Table 8.1, 8.2 and 8.3). The lack of temperature differences between unlogged and logged forests suggests that thermal regimes between stream types were not different

Table 8.1. Analysis of Variance (General Linear Model), significance test for water temperature in the streams flowing through the Unlogged and Logged sites in different seasons at Camp 92, Central Kalimantan

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Large Stream						
Wet 1						
Site	1	0.15	0.15	0.15	1.47	0.243
Measurement	3	0.14	0.14	0.04	0.48	0.700
Site*Measure	3	0.21	0.21	0.07	0.69	0.573
Error	16	1.64	1.64	0.10		
Total	23	2.14				
Wet 2						
Site	1	0.08	0.08	0.08	0.75	0.398
Measurement	3	0.20	0.20	0.06	0.64	0.600
Site*Measure	3	0.19	0.19	0.06	0.60	0.624
Error	16	1.73	1.73	0.10		
Total	23	2.21				
Dry						
Site	1	0.40	0.40	0.40	3.10	0.097
Measurement	3	0.60	0.60	0.20	1.57	0.236
Site*Measure	3	0.28	0.28	0.09	0.73	0.547
Error	16	2.06	2.06	0.13		
Total	23	3.36				

Table ... contd.

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Small Stream						
Wet 1						
Site	1	0.24	0.24	0.24	1.29	0.272
Measurement	3	0.84	0.84	0.28	1.52	0.248
Site*Measure	3	0.09	0.09	0.03	0.17	0.917
Error	16	2.96	2.96	0.18		
Total	23	4.14				
Wet 2						
Site	1	0.006	0.006	0.006	0.04	0.853
Measurement	3	0.44	0.44	0.14	0.78	0.521
Site*Measure	3	0.96	0.96	0.32	1.72	0.203
Error	16	3.00	3.00	0.18		
Total	23	4.41				
Dry						
Site	1	0.20	0.20	0.20	0.55	0.470
Measurement	3	0.25	0.25	0.08	0.23	0.873
Site*Measure	3	0.01	0.01	0.005	0.01	0.998
Error	16	5.88	5.88	0.36		
Total	23	6.35				

Table 8.2. Analysis of Variance (General Linear Model), significance test for stream depth in the surveyed section of the Unlogged and Logged sites in different seasons at Camp 92, Central Kalimantan

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Large Stream						
Wet 1						
Site	1	0.08	0.08	0.08	8.45	0.007*
Error	28	0.28	0.28	0.01		
Total	29	0.36				
Wet 2						
Site	1	0.17	0.17	0.17	7.65	0.01**
Error	28	0.64	0.64	0.02		
Total	29	0.82				
Dry						
Site	1	0.17	0.17	0.17	43.56	0.000***
Error	28	0.11	0.004			
Total	29	0.29				
Small Stream						
Wet 1						
Site	1	0.001	0.001	0.001	0.25	0.618
Error	28	0.14	0.14	0.005		
Total	29	0.14				
Wet 2						
Site	1	0.005	0.005	0.005	0.97	0.332
Error	28	0.15	0.15	0.005		
Total	29	0.15				
Dry						
Site	1	0.0003	0.0003	0.0003	0.05	0.833
Error	28	0.20	0.20	0.007		
Total	29	0.20				

The large streams have permanent water supplies which flow continuously throughout the season, whilst the small streams have impermanent water supplies and are largely dependent on rainfall events to fill the stream beds. Therefore, the differences of water depths between the unlogged and logged plots were significant in the large streams.

Streams flowing through Unlogged Plots had significantly different % canopy cover than the Logged Plots (Table 8.3).

Table 8.3. Analysis of Variance (General Linear Model), significance test for % canopy cover in the surveyed section of the Unlogged and Logged sites in different seasons at Camp 92, Central Kalimantan

Source	Df	Seq SS	Adj SS	Adj MS	F	P
Large Stream						
Wet season						
Site	1	642.00	642.00	642.00	45.85	0.000***
Error	28	392.00	392.05	14.00		
Total	29	1034.05				
Dry season						
Site	1	521.92	521.92	521.92	15.95	0.000***
Error	28	916.49	916.49	32.73		
Total	29	1438.41				
Small Stream						
Wet season						
Site	1	623.99	623.99	623.99	53.20	0.000***
Error	28	328.39	328.39	11.73		
Total	29	952.38				
Dry season						
Site	1	484.73	484.73	484.73	25.22	0.000***
Error	28	538.06	538.06	19.22		
Total	29	1022.80				

This resulted from dense stands of dipterocarps along most primary forest streams in the research sites. In contrast, canopy cover shading over the streams flowing through the logged stands was less than in the unlogged stands because logging reduced the canopy cover to some extent and created the canopy gaps. A significant difference also existed in the distribution of substrates between streams in the unlogged and the logged areas (Fig 8.1 and 8.2). A greater number of the observed

substrates in the streams flowing through the logged areas were of the smallest size classes (silt, sand, and gravel), whereas the coarse substrates (cobble and boulder) were more common in the unlogged areas ($P < 0.001$).

Fig 8.1 Distribution of substrates for small streams in uncut forests and streams in the logged stands at Kalang Forest Stream, Central Kalimantan

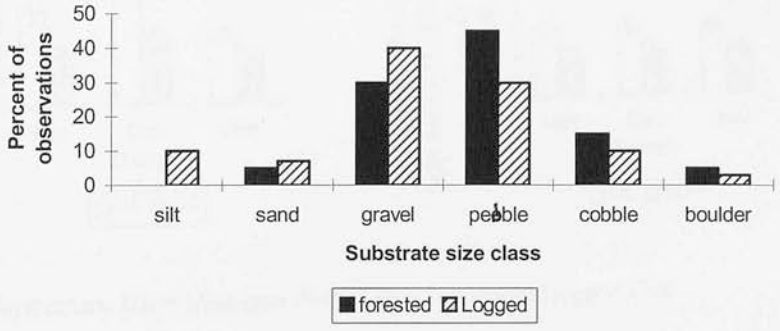
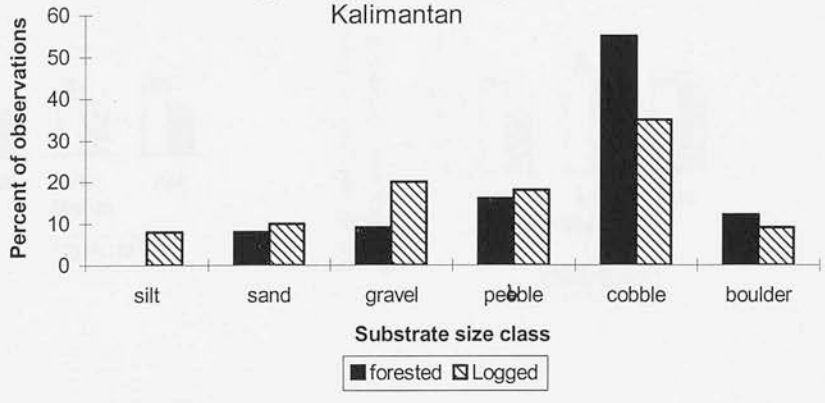


Fig 8.2 Distribution of substrates for large streams in uncut forests and streams in the logged stands at Kalang Forest Stream, Central Kalimantan



8.4. Abundance of Anurans

8.4.1. *Differences in anuran populations*

The number of individual anurans observed per night transect in the small and large forest streams showed a variation between seasons. The average numbers of individual anurans in the small and large streams of primary forests (Unlogged

Plots) were much higher than in the logged forests (Figure 8.3, 8.4, 8.5, 8.6, 8.7 and 8.8).

Fig 8.3 Number of individual anurans during wet season 1 in different sites at Kalang Small Forest Stream, Central Kalimantan

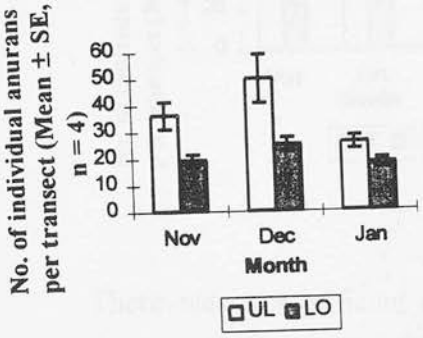
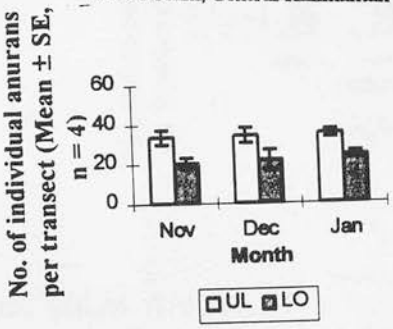


Fig 8.4 Number of individual anurans during wet season 1 in different sites at Kalang Large Forest Stream, Central Kalimantan



Bars are standard errors UL = Unlogged Plot LO = 2-year-old Logged Plot

Fig 8.5 Number of individual anurans during wet season 2 in different sites at Kalang Small Forest Stream, Central Kalimantan

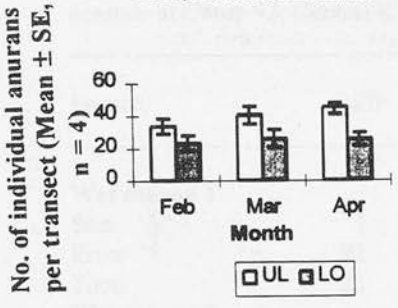


Fig 8.6 Number of individual anurans during wet season 2 in different sites at Kalang Large Forest Stream, Central Kalimantan

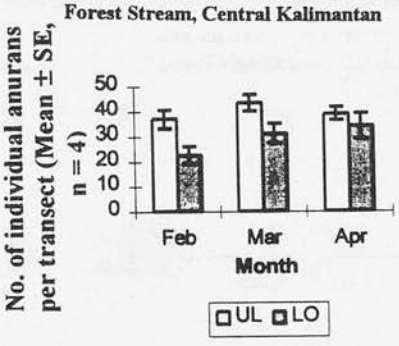


Fig 8.7 Number of individual anurans during dry season in different sites at Kalang Small Forest Stream, Central Kalimantan

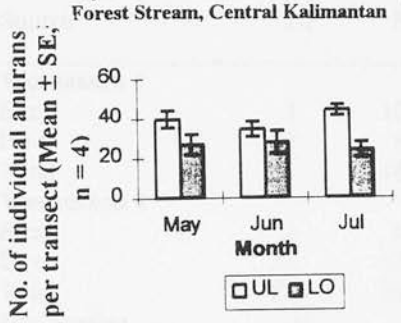
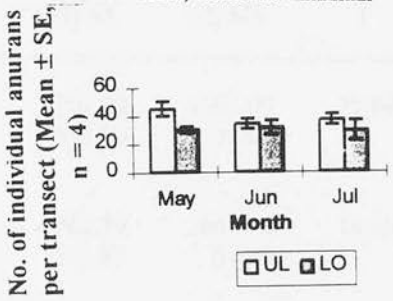


Fig 8.8 Number of individual anurans during dry season in different sites at Kalang Large Forest Stream, Central Kalimantan



There was a significant difference (GLM ANOVA, $P < 0.05$) in abundance of anurans per stream transect between streams flowing through unlogged forests and streams flowing through 2-year-old logged forests (Table 8.4 and 8.5).

Table 8.4 Analysis of Variance (General Linear Model), significance test for abundance of anurans between small streams flowing through unlogged forest and 2-year-old logged forest in different seasons at Camp 92, Central Kalimantan

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Wet season 1						
Site	1	1633.50	1633.50	1633.50	12.67	0.002**
Error	22	2836.30	2836.30	128.90		
Total	23	4469.80				
Wet season 2						
Site	1	1320.20	1320.20	1320.20	16.81	0.000***
Error	22	1727.70	1727.70	78.50		
Total	23	3047.80				
Dry Season						
Site	1	1040.20	1040.20	1040.20	14.50	0.001***
Error	22	1577.70	1577.70	71.70		
Total	23	2617.80				

Table 8.5 Analysis of Variance (General Linear Model), significance test for abundance of anurans between large streams flowing through unlogged forest and 2-year-old logged forest in different seasons at Camp 92, Central Kalimantan

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Wet season 1						
Site	1	1001.00	1001.00	1001.00	25.64	0.000***
Error	22	858.90	858.90	39.00		
Total	23	1860.00				
Wet season 2						
Site	1	661.50	661.50	661.50	16.69	0.004**
Error	22	1361.83	1361.83	61.90		
Total	23	2023.33				
Dry Season						
Site	1	345.04	345.04	345.04	8.76	0.006**
Error	22	2020.92	2020.92	91.86		
Total	23	2365.96				

8.4.2. *Abundance of anurans in relation to humidity*

Table 8.6. Analysis of Covariance for abundance of anurans in relation to humidity between seasons in Kalang Small Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Humidity	3	2312.11	770.70	27.93	0.000***
Site	1	1.98	1.98	0.07	0.792
Error	19	524.22	27.59		
Total	23	4469.83			
Wet season 2					
Humidity	3	1315.41	438.47	20.21	0.000***
Site	1	0.04	0.04	0.02	0.968
Error	19	412.26	21.70		
Total	23	3047.83			
Dry season					
Humidity	3	1371.10	457.03	42.04	0.000***
Site	1	76.71	76.71	7.06	0.016*
Error	19	206.57	10.87		
Total	23	2617.83			

Table 8.6 shows that humidity had a very highly significant (ANCOVA, $P < 0.001$) effect on abundance of anurans in Kalang small forest streams. Regression analysis was used to describe the relationship between humidity and abundance of anurans.

Figure 8.9 Regression analysis for humidity and abundance of anurans during wet season 1 in Kalang small forest stream at Camp 92, Central Kalimantan

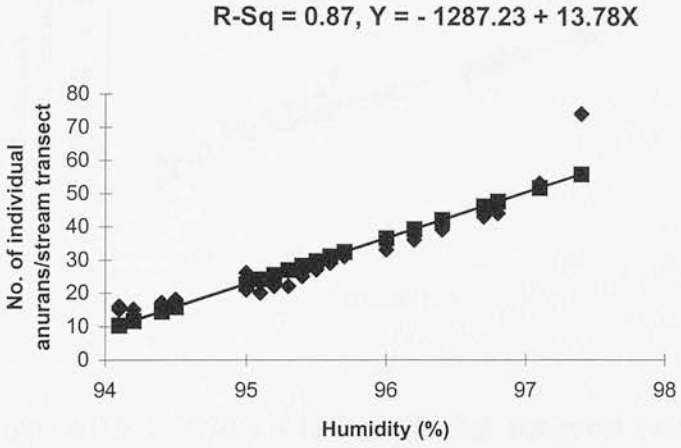


Figure 8.10 Regression analysis for humidity and abundance of anurans during wet season 2 in Kalang small forest stream at Camp 92, Central Kalimantan

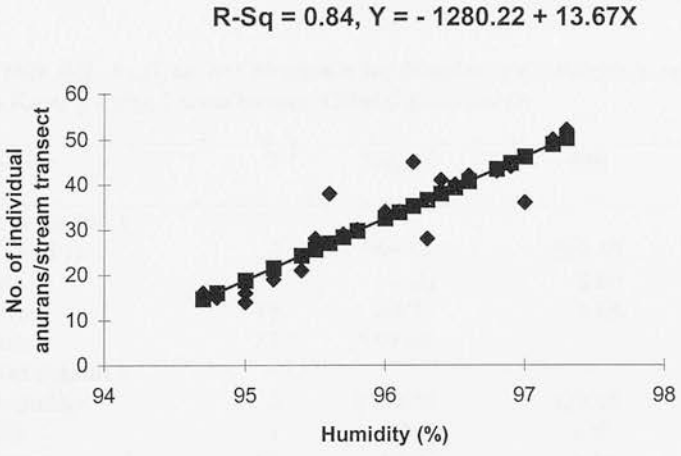
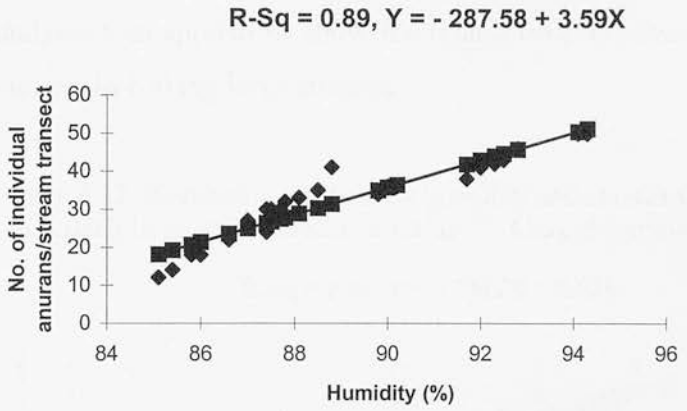


Figure 8.11 Regression analysis for humidity and abundance of anurans during dry season in Kalang small forest stream at Camp 92, Central Kalimantan



Figures 8.9, 8.10 and 8.11 indicate that statistical evidence (R-Sq = 0.87, 0.84 and 0.89, p-value < 0.0001, n = 24) strongly supported the hypothesis that the number of anurans captured is related to humidity. The regression analysis indicates a positive relationship between variables in which increased humidity leads to increased abundance of anurans.

Table 8.7 Analysis of Covariance for abundance of anurans in relation to humidity between seasons in Kalang Large Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Humidity	3	789.58	263.19	72.13	0.000***
Site	1	2.00	2.00	0.55	0.469
Error	19	69.33	3.65		
Total	23	1859.96			
Wet season 2					
Humidity	3	1281.21	427.07	100.65	0.000***
Site	1	10.36	10.36	2.44	0.135
Error	19	80.62	4.24		
Total	23	2023.33			
Dry season					
Humidity	3	1743.41	581.14	39.79	0.000***
Site	1	11.85	11.85	0.81	0.379
Error	19	277.50	14.61		
Total	23	2365.96			

Table 8.7 shows humidity also had a very highly significant (ANCOVA, $P < 0.001$) effect on abundance of anurans recorded in Kalang large forest streams. Regression analysis was applied to show the relationship between humidity and abundance of anurans in Kalang large streams.

Figure 8.12 Regression analysis for humidity and abundance of anurans during wet season 1 in Kalang large forest stream at Camp 92, Central Kalimantan

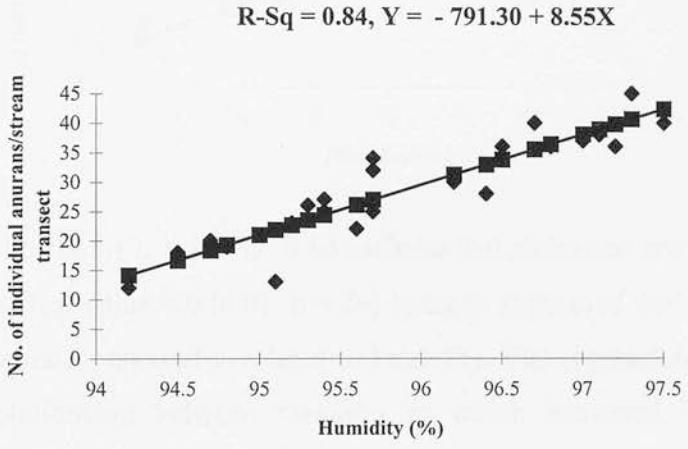


Figure 8.13 Regression analysis for humidity and abundance of anurans during wet season 2 in Kalang large forest stream at Camp 92, Central Kalimantan

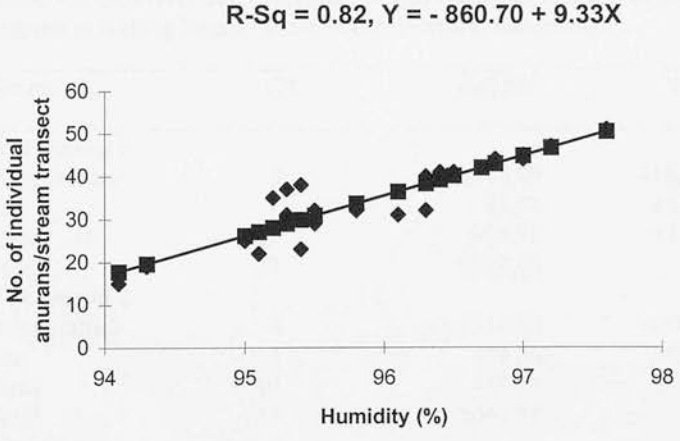
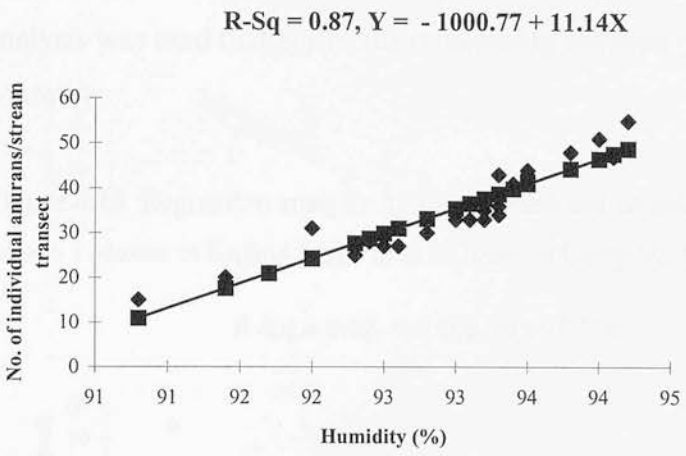


Figure 8.14 Regression analysis for humidity and abundance of anurans during dry season in Kalang large forest stream at Camp 92, Central Kalimantan



Figures 8.12, 8.13 and 8.14 indicate that statistical evidence (R-Sq = 0.84, 0.82 and 0.87, p-value < 0.0001, n = 24) strongly supported the hypothesis that the number of anurans captured is related to humidity. The regression analysis indicates a positive relationship between variables in which increased humidity leads to increased abundance of anurans.

8.4.3. *Abundance of Anurans in relation to temperature*

Table 8.8 Analysis of Covariance for abundance of anurans in relation to temperature between seasons in Kalang Small Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Temperature	3	1248.89	416.30	24.06	0.000***
Site	1	43.05	43.05	2.49	0.131
Error	19	328.78	17.30		
Total	23	2617.83			
Wet season 2					
Temperature	3	1312.50	437.50	20.02	0.000***
Site	1	24.59	24.59	1.13	0.302
Error	19	415.17	21.85		
Total	23	3047.83			
Dry season					
Temperature	3	2345.16	781.03	30.24	0.000***
Site	1	292.01	292.01	11.30	0.003**
Error	19	491.17	25.85		
Total	23	4469.83			

Table 8.8 shows that temperature had a very highly significant (ANCOVA, P <0.001) effect on abundance of anurans in Kalang small forest streams. Regression analysis was used to describe the relationship between temperature and abundance of anurans.

Figure 8.15 Regression analysis for temperature and abundance of anurans during wet season 1 season in Kalang small forest stream at Camp 92, Central Kalimantan

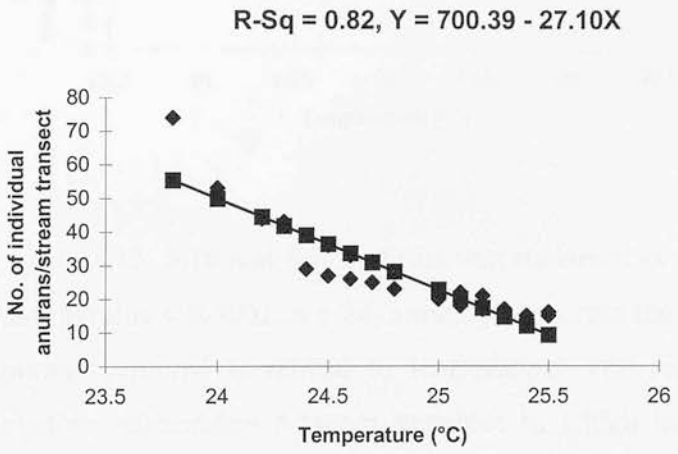


Figure 8.16 Regression analysis for temperature and abundance of anurans during wet season 2 season in Kalang small forest stream at Camp 92, Central Kalimantan

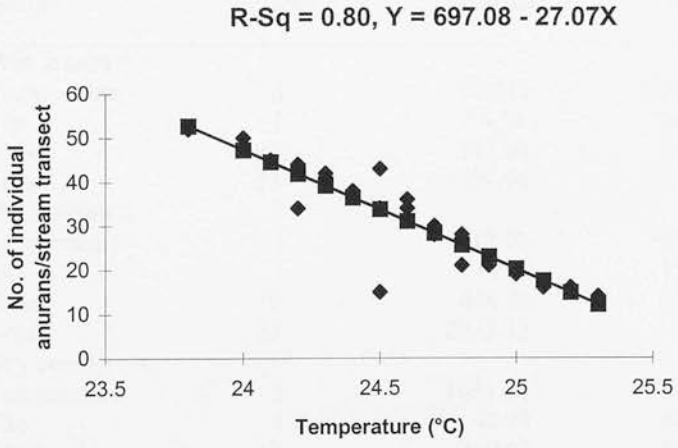
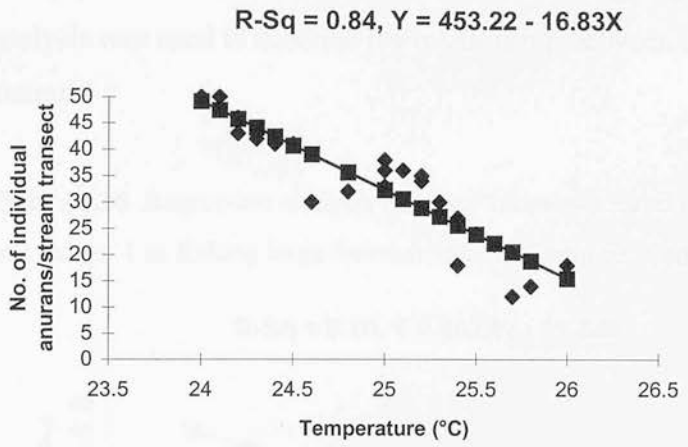


Figure 8.17 Regression analysis between temperature and abundance of anurans during dry season season in Kalang small forest stream at Camp 92, Central Kalimantan



Figures 8.15, 8.16 and 8.17 indicate that statistical evidence (R-Sq = 0.82, 0.80 and 0.84, p-value < 0.0001, n = 24) strongly supported the hypothesis that the number of anurans captured is related to temperature. The regression analysis indicates a negative relationship between variables in which increased temperature leads to decreased abundance of anurans.

Table 8.9 Analysis of Covariance for abundance of anurans in relation to temperature between seasons in Kalang Large Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Temperature	3	717.11	239.04	32.03	0.000***
Site	1	76.24	76.24	3.61	0.100
Error	19	141.81	7.46		
Total	23	1859.96			
Wet season 2					
Temperature	3	917.29	305.76	13.07	0.000***
Site	1	19.39	19.39	0.83	0.374
Error	19	444.55	23.40		
Total	23	2023.33			
Dry season					
Temperature	3	1651.88	550.63	28.35	0.000***
Site	1	42.44	42.44	2.19	0.156
Error	19	369.03	19.42		
Total	23	2365.96			

Table 8.9 shows that temperature had a very highly significant (ANCOVA, P <0.001) effect on abundance of anurans in Kalang large forest streams. Regression analysis was used to describe the relationship between temperature and abundance of anurans.

Figure 8.18 Regression analysis between temperature and abundance of anurans during wet season 1 in Kalang large forest stream at Camp 92, Central Kalimantan

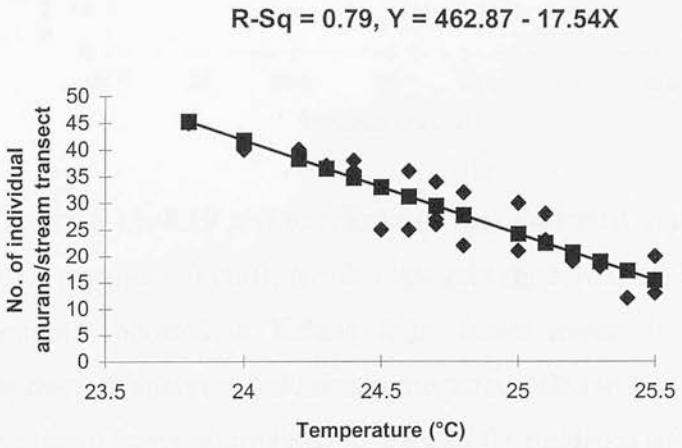


Figure 8.19 Regression analysis between temperature and abundance of anurans during wet season 2 season in Kalang large forest stream at Camp 92, Central Kalimantan

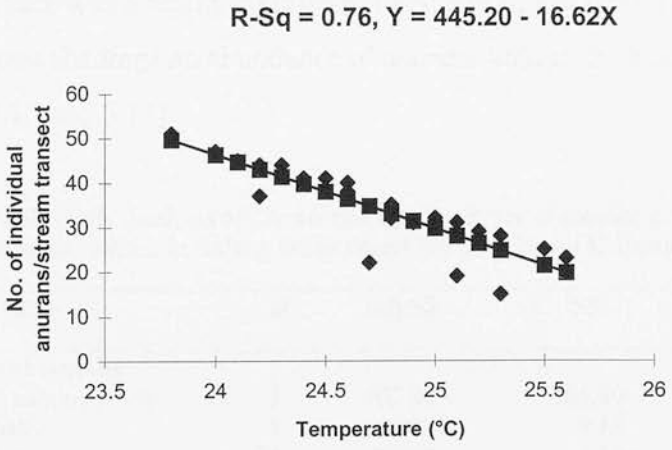
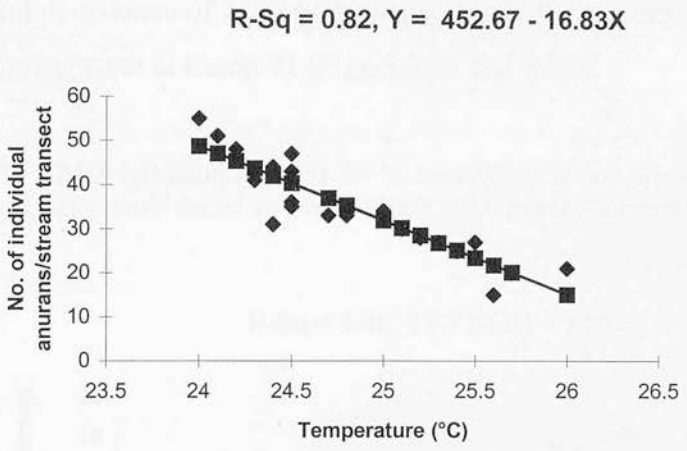


Figure 8.20 Regression analysis between temperature and abundance of anurans during dry season in Kalang large forest stream at Camp 92, Central Kalimantan



Figures 8.18, 8.19 and 8.20 indicate that statistical evidence (R-Sq = 0.79, 0.76 and 0.82, p-value < 0.0001, n = 24) strongly supported the hypothesis that the number of anurans captured in Kalang large forest stream is related to temperature. The regression analysis indicates a negative relationship between variables in which increased temperature leads to decreased abundance of anurans.

8.4.4. *Vegetation and Abundance of Anurans*

There was a highly significant (ANCOVA, $P < 0.001$) effect of percent canopy cover (tree shading) on abundance of anurans both in the small and large streams (Tables 8.10 and 8.11).

Table 8.10 Analysis of Covariance for abundance of anurans in relation to % canopy cover between seasons in Kalang Small Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet seasons					
% canopy cover	1	505.80	505.80	150.29	0.000***
Site	1	9.14	9.14	2.72	0.111
Error	27	90.87	3.37		
Total	29	1903.47			
Dry season					
% canopy cover	1	908.65	908.65	96.18	0.000***
Site	1	3.27	3.27	0.35	0.561
Error	27	255.09	9.45		
Total	29	1995.87			

Regression analysis was used to describe the relationship between % canopy cover and abundance of anurans during wet and dry seasons in the small forest streams of Kalang river at Camp 92 (Figure 8.21 and 8.22).

Fig 8.21 Regression analysis for % canopy cover and abundance of anurans (wet seasons) at Kalang small forest stream at Camp 92, Central Kalimantan

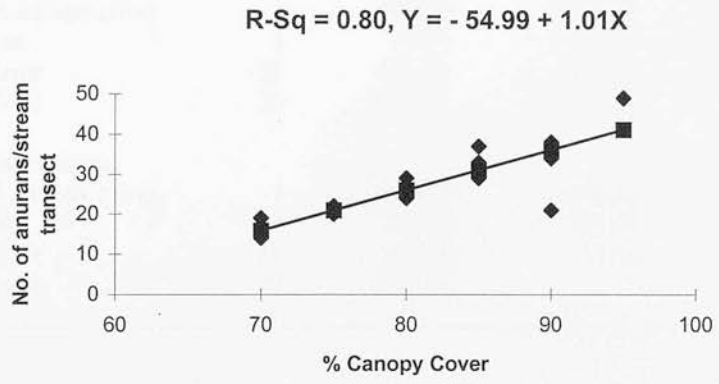
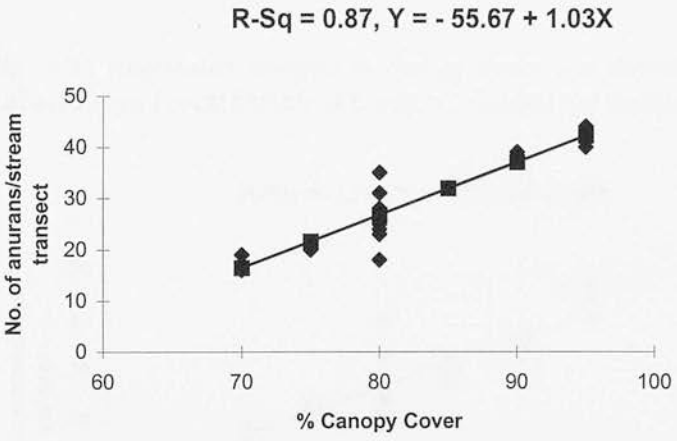


Fig 8.22 Regression analysis for % canopy cover and abundance of anurans (dry season) at Kalang small forest stream at Camp 92, Central Kalimantan



The values of R-Sq at 0.80 (wet season) and 0.87 (dry season) show that more than 80% of the variability in the data set has been accounted for by the regression model. The statistical evidence (R-Sq = 0.80 and 0.87, p-value < 0.0001, n = 30) strongly supported the hyphothesis that the number of anurans captured is related to %

canopy cover in wet and dry seasons. The regression analysis shows a positive relationship in which increased % canopy cover leads to increased number of anurans captured.

Table 8.11 Analysis of Covariance for abundance of anurans in relation to % canopy cover between seasons in Kalang Large Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
% canopy cover	1	900.61	900.61	85.64	0.000***
Site	1	38.70	38.70	3.68	0.066
Error	27	283.92	10.52		
Total	29	1985.37			
Dry season					
% canopy cover	1	2368.20	2368.20	149.43	0.000***
Site	1	282.30	282.30	17.81	0.010**
Error	27	427.90	15.80		
Total	29	3072.20			

Regression analysis was used to describe the relationship between % canopy cover and abundance of anurans during wet and dry seasons in the large forest streams of Kalang river (Figures 8.23 and 8.24)

Fig 8.23 Regression analysis % canopy cover and abundance of anurans (wet season) at Kalang Large Forest Stream at Camp 92, Central Kalimantan

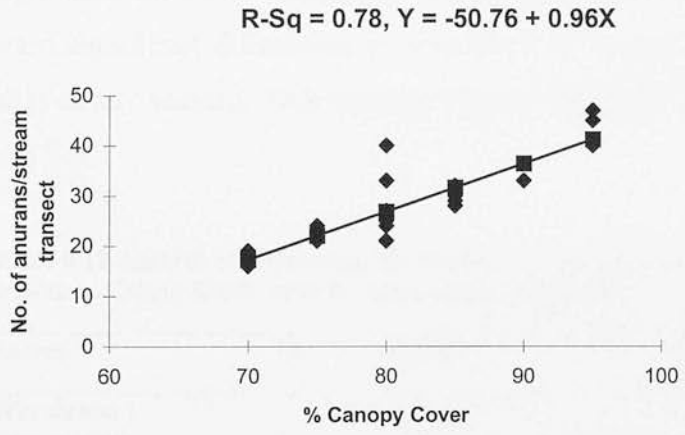
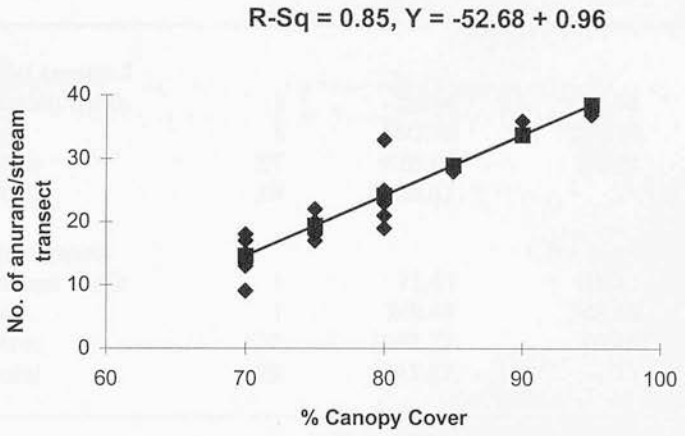


Fig 8.24 Regression analysis for % canopy cover and abundance of anurans (dry season) at Kalang Large Forest Streams at Camp 92, Central Kalimantan



The values of R-Sq at 0.78 (wet season) and 0.82 (dry season) showed that more than 80% of the variability in the data set has been accounted for by the regression model. The statistical evidence (R-Sq = 0.78 and 0.82, p-value < 0.0001, n = 30) strongly supports the hyphothesis that the number of anurans captured is related to % canopy cover in wet and dry seasons. The regression analysis shows a positive relationship in which increased % canopy cover leads to increased number of anurans captured.

8.4.5. *Stream Width and Abundance of Anurans*

Table 8.12 and 8.13 show that stream widths in the small and large streams had no significant effect on abundance of anurans (ANCOVA, P > 0.05). However, there were significant differences in abundance of anurans in the unlogged and logged sites at any seasons. This strongly support the significance test results in Table 8.4 and 8.5.

Table 8.12 Analysis of Covariance for abundance of anurans in relation to stream width between seasons in Kalang Small Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Stream width	1	30.30	30.30	1.43	0.242
Site	1	1263.19	1263.19	60.19	0.000***
Error	27	566.66	20.99		
Total	29	1903.47			

Table...contd.

Source	Df	Adj SS	MS	F	P
Wet season 2					
Stream width	1	83.44	83.44	2.53	0.123
Site	1	997.78	997.78	30.30	0.000***
Error	27	889.09	32.93		
Total	29	2028.67			
Dry season					
Stream width	1	71.47	71.47	1.77	0.195
Site	1	748.48	748.48	19.39	0.000***
Error	27	1092.27	40.45		
Total	29	1995.87			

Table 8.13 Analysis of Covariance for abundance of anurans in relation to stream width between seasons in Kalang Large Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Stream width	1	2.21	2.21	0.05	0.824
Site	1	677.07	677.07	15.46	0.001***
Error	27	1182.32	43.79		
Total	29	1985.37			
Wet season 2					
Stream width	1	1.19	1.19	0.02	0.877
Site	1	449.14	449.14	9.24	0.005**
Error	27	1313.07	48.63		
Total	29	1843.47			
Dry season					
Stream width	1	15.30	15.30	0.15	0.703
Site	1	602.20	602.20	1 1.96	0.003 **
Error	27	2780.80	40.45		
Total	29	3072.20			

8.4.6. *Stream Depth and Abundance of Anurans*

Table 8.14 and 8.15 show that stream depths in the small and large streams during dry and wet seasons had no significant effect on the abundance of anurans. But again there were significant differences in abundance of anurans between sites at any

seasons. This is strongly support the results of significance tests in Table 8.4 and 8.5.

Table 8.14 Analysis of Covariance for abundance of anurans in relation to stream depth between seasons in Kalang Small Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Stream depth	1	63.84	63.84	3.24	0.083
Site	1	1241.02	1241.02	62.89	0.000***
Error	27	532.82	19.73		
Total	29	1903.47			
Wet season 2					
Stream depth	1	41.11	41.11	1.19	0.285
Site	1	666.03	666.03	19.31	0.000***
Error	27	931.43	34.50		
Total	29	2028.67			
Dry season					
Stream depth	1	102.09	102.09	2.60	0.119
Site	1	807.58	807.58	20.54	0.000***
Error	27	1061.27	39.32		
Total	29	1995.87			

Table 8.15 Analysis of Covariance for abundance of anurans in relation to stream depth between seasons in Kalang Large Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Stream depth	1	74.16	74.16	1.80	0.191
Site	1	504.06	504.06	12.26	0.002**
Error	27	1110.38	41.13		
Total	29	1985.37			
Wet season 2					
Stream depth	1	50.96	50.96	1.09	0.306
Site	1	579.71	579.71	12.39	0.002**
Error	27	1263.30	46.79		
Total	29	1843.47			
Dry season					
Stream depth	1	484.93	484.93	5.67	0.125
Site	1	699.04	699.04	8.17	0.008**
Error	27	2311.20	85.60		
Total	29	3072.20			

8.5. Anuran Species Richness

8.5.1. *Differences in numbers of anuran species*

The number of anuran species observed per night transect varied slightly between seasons in different sites (small and large forest streams) as shown in Figures 8.25, 8.26, 8.27, 8.28, 8.29 and 8.30.

Fig 8.25 Mean number of anuran species observed during wet season 1 in different sites at Kalang Small Forest Stream at Camp 92, Central Kalimantan

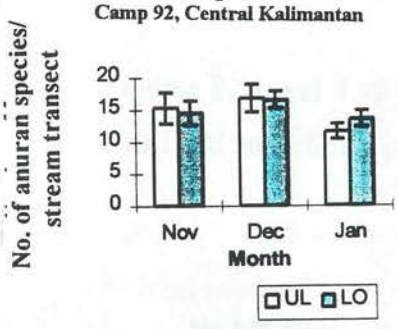
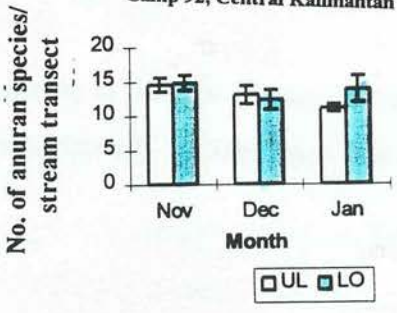


Fig 8.26 Mean number of anuran species observed during wet season 1 in different sites at Kalang Large Forest Stream at Camp 92, Central Kalimantan



Bars are standard errors UL = Unlogged Plot LO = 2-year-old Logged Plot

Fig 8.27 Mean number of anuran species observed during wet season 2 in different sites at Kalang Small Forest Stream at Camp 92, Central Kalimantan

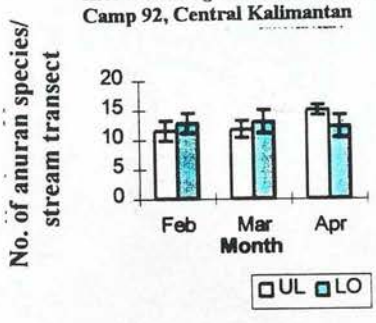


Fig 8.28 Mean number of anuran species observed during wet season 2 in different sites at Kalang Large Forest Stream at Camp 92, Central Kalimantan

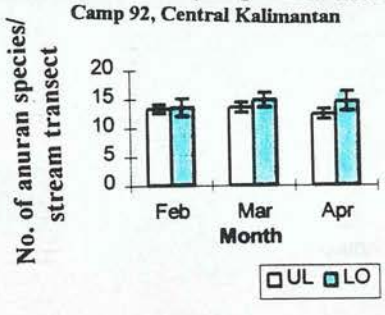


Fig 8.29 Mean number of anuran species observed during the dry season in different sites at Kalang Small Forest Stream at Camp 92, Central Kalimantan

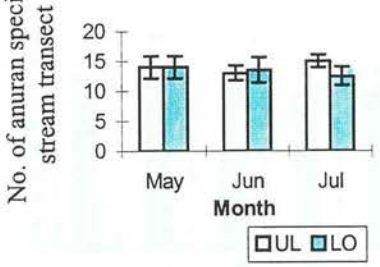
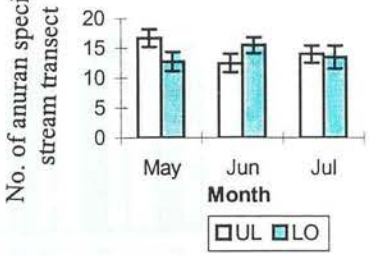
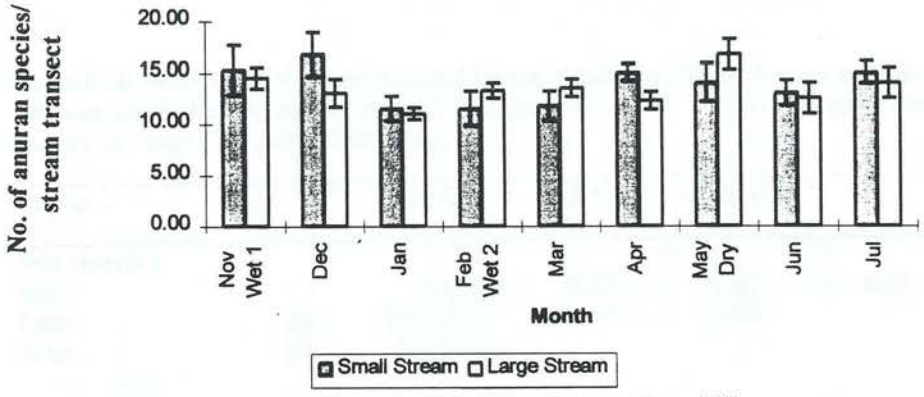


Fig 8.30 Mean number of anuran species observed during the dry season in different sites at Kalang Large Forest Stream at Camp 92, Central Kalimantan



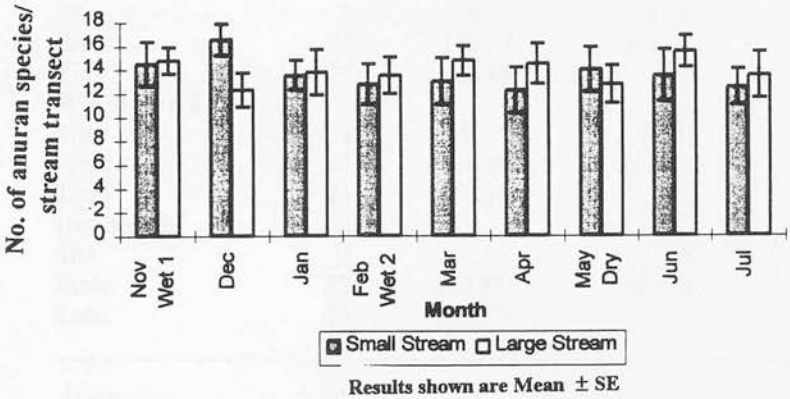
Figures 8.31 and 8.32 show a variation in species richness between seasons in the small and large forest streams flowing through the Unlogged and Logged forests.

Fig 8.31 Mean number of anuran species between seasons in different forest streams flowing through the unlogged forest at Camp 92, Central Kalimantan



Results shown are Mean ± SE

Fig 8.32 Mean number of anuran species between seasons in different forest streams flowing through the 2-year-old logged forest at Camp 92, Central Kalimantan



Although there was a variation in species richness between seasons in different sample streams, they were not statistically significant (GLM ANOVA, $P > 0.05$). Table 8.16 and 8.17 show there was no significant difference between the average number of anuran species/stream transect between seasons in the Unlogged and Logged forests.

Table 8.16 Analysis of Variance (General Linear Model), significant test for anuran species richness between small streams flowing through unlogged forest and 2-year-old logged forest in different seasons at Camp 92, Central Kalimantan

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Wet season 1						
Site	1	0.67	0.67	0.67	0.05	0.828
Error	22	302.67	302.67	13.76		
Total	23	303.33				
Wet season 2						
Site	1	0.04	0.04	0.04	0.01	0.950
Error	22	224.92	224.92	10.22		
Total	23	224.96				
Dry Season						
Site	1	2.66	2.66	2.66	0.28	0.603
Error	22	210.66	210.66	9.57		
Total	23	213.33				

Table 8.17 Analysis of Variance (General Linear Model), significant test for anuran species richness between large streams flowing through unlogged forest and 2-year-old logged forest in different seasons at Camp 92, Central Kalimantan

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Wet season 1						
Site	1	3.37	3.37	3.37	0.46	0.504
Error	22	160.58	160.58	7.30		
Total	23	163.96				
Wet season 2						
Site	1	9.37	9.37	9.37	1.81	0.193
Error	22	114.25	114.25	5.19		
Total	23	123.62				
Dry Season						
Site	1	1.50	1.50	1.50	0.14	0.708
Error	22	229.83	229.83	10.45		
Total	23	231.33				

8.5.2. *Humidity and Species Richness*

Table 8.18 shows that humidity had no significant effect (ANCOVA, $P > 0.05$, $n = 24$) on species richness in the small Kalang streams during the wet seasons but there was a significant effect (ANCOVA, $P < 0.01$) of humidity on species richness during the dry season.

Table 8.18 Analysis of Covariance for anuran species in relation to humidity between seasons in Kalang Small Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Humidity	3	52.18	17.39	1.32	0.297
Site	1	14.51	14.51	1.10	0.307
Error	19	250.49	13.18		
Total	23	303.33			
Wet season 2					
Humidity	3	68.72	22.91	2.79	0.069
Site	1	9.65	9.65	1.17	0.292
Error	19	156.19	8.22		
Total	23	224.96			
Dry season					
Humidity	3	94.27	31.42	5.13	0.009**
Site	1	11.05	11.05	1.80	0.195
Error	19	116.39	6.12		
Total	23	213.33			

Table 8.19 also shows that humidity had a significant effect (ANCOVA, P > 0.05, n = 24) on species richness in the large Kalang streams only in the dry season.

Table 8.19 Analysis of Covariance for anuran species in relation to humidity between seasons in Kalang Large Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Humidity	3	52.18	17.39	1.32	0.297
Site	1	14.51	14.51	1.10	0.307
Error	19	250.49	13.18		
Total	23	303.33			
Wet season 2					
Humidity	3	68.72	22.91	2.79	0.069
Site	1	9.65	9.65	1.17	0.292
Error	19	156.19	8.22		
Total	23	224.96			
Dry season					
Humidity	3	94.27	31.42	5.13	0.009**
Site	1	11.05	11.05	1.80	0.195
Error	19	116.39	6.12		
Total	23	213.33			

8.5.3. *Percent Canopy Cover and Species Richness*

There was no significant effect of canopy cover (% tree shading on stream banks) on anuran species richness during the wet season in the small and large of Kalang Forest Streams. However, there was a significant effect of canopy cover on anuran species richness during the dry season in streams of both sizes in Kalang Forest (Table 8.20 and 8.21).

Table 8.20 Analysis of Covariance for anuran species in relation to % canopy cover between seasons in Kalang Large Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
% canopy cover	1	4.55	4.55	3.62	0.068
Site	1	1.25	1.25	1.00	0.327
Error	27	34.98	1.26		
Total	29	39.36			
Dry season					
% canopy cover	1	10.66	10.66	13.58	0.001**
Site	1	2.50	2.50	3.19	0.085
Error	27	21.20	0.85		
Total	29	31.86			

Table 8.21 Analysis of Covariance for anuran species in relation to % canopy cover between seasons in Kalang Small Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
% canopy cover	1	3.25	3.25	2.80	0.106
Site	1	2.61	2.61	2.24	0.146
Error	27	31.41	1.16		
Total	29	45.46			
Dry season					
% canopy cover	1	6.60	6.60	9.26	0.005**
Site	1	2.89	2.89	2.16	0.068
Error	27	19.26	0.71		
Total	29	26.70			

8.5.4. *Stream Width and Species Richness*

Table 8.22 and 8.23 show that stream widths had no significant effect on the number of anuran species both in small and large streams in any season (ANCOVA, $P > 0.05$).

Table 8.22 Analysis of Covariance for abundance of anurans in relation to stream width between seasons in Kalang Small Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Stream width	1	0.07	0.07	0.06	0.814
Site	1	10.87	10.87	8.49	0.107
Error	27	34.59	1.28		
Total	29	45.46			
Wet season 2					
Stream width	1	2.16	2.16	4.58	0.081
Site	1	0.03	0.03	0.01	0.937
Error	27	12.76	0.47		
Total	29	14.96			
Dry season					
Stream width	1	0.25	0.25	0.27	0.607
Site	1	0.90	0.90	0.96	0.336
Error	27	25.61			
Total	29	26.70			

Table 8.23 Analysis of Covariance for anuran species in relation to stream width between seasons in Kalang Large Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Stream width	1	0.03	0.03	0.02	0.883
Site	1	0.63	0.63	0.44	0.511
Error	27	38.50	1.42		
Total	29	39.36			
Wet season 2					
Stream width	1	1.17	1.17	1.09	0.305
Site	1	0.30	0.30	0.28	0.602
Error	27	28.96	1.07		
Total	29	30.16			
Dry season					
Stream width	1	0.35	0.35	0.30	0.588
Site	1	0.04	0.04	0.04	0.850
Error	27	31.51	1.16		
Total	29	31.86			

8.5.5. *Stream depth and species richness*

Stream depth in different sites also had no significant effect (ANCOVA, $P > 0.05$) on species richness in any season (Table 8.24 and 8.25).

Table 8.24 Analysis of Covariance for anuran species in relation to stream depth between seasons in Kalang Small Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Stream depth	1	0.07	0.07	0.05	0.819
Site	1	10.86	10.86	8.48	0.107
Error	27	34.59	1.28		
Total	29	45.46			
Wet season 2					
Stream depth	1	0.26	0.26	0.49	0.489
Site	1	0.16	0.16	0.30	0.590
Error	27	14.66	0.54		
Total	29	14.96			
Dry season					
Stream depth	1	0.16	0.16	0.18	0.677
Site	1	0.86	0.86	0.91	0.350
Error	27	25.69	0.95		
Total	29	26.70			

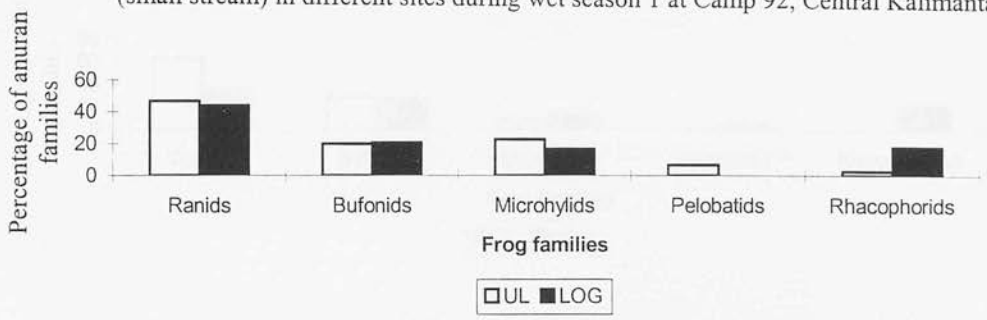
Table 8.25 Analysis of Covariance for abundance of anurans in relation to stream depth between seasons in Kalang Large Forest Stream, Central Kalimantan

Source	Df	Adj SS	MS	F	P
Wet season 1					
Stream depth	1	2.22	2.22	1.66	0.209
Site	1	0.06	0.06	0.04	0.844
Error	27	36.30	1.34		
Total	29	39.36			
Wet season 2					
Stream depth	1	0.47	0.47	0.43	0.519
Site	1	0.15	0.15	0.14	0.710
Error	27	29.66	1.10		
Total	29	30.16			
Dry season					
Stream depth	1	0.37	0.37	0.32	0.574
Site	1	0.23	0.23	0.20	0.661
Error	27	31.49	1.16		
Total	29	31.86			

8.6. Anuran Species Composition

A total of 43 species was observed from small and large forest streams flowing through the Unlogged and Logged forests at Camp 92, Central Kalimantan (see Table 2.1). The proportion of anuran species grouped by family recorded from night riparian transects between seasons in the small and large streams is presented in Figures 8.33 to 8.38. Of the total number of species recorded from small and large forest streams flowing through the unlogged and logged plots, the largest proportion was composed of ranids. Bufonids were the second largest group and pelobatids were the smallest group. Rhachophorids were observed particularly in the logged plots and rarely in the primary forest streams. The observation of some species of tree anurans (Rhacophorids) in the logged areas may indicate that this species had moved from their original habitats of pristine forest. This could possibly have been an effect of logging in their main habitat.

Fig. 8.33 The proportion of individual anurans in each family recorded from night riparian transects (small stream) in different sites during wet season 1 at Camp 92, Central Kalimantan



UL = Unlogged Plot LOG = 2-year-old Logged Plot

Fig. 8.34 The proportion of individual anurans in each family recorded from night riparian transects (small stream) in different sites during wet season 2 at Camp 92, Central Kalimantan

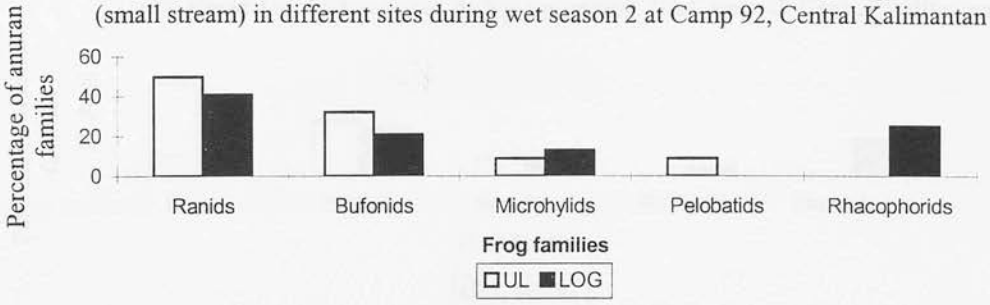


Fig. 8.35 The proportion of individual anurans in each family recorded from night riparian transects (small stream) in different sites during the dry season at Camp 92, Central Kalimantan

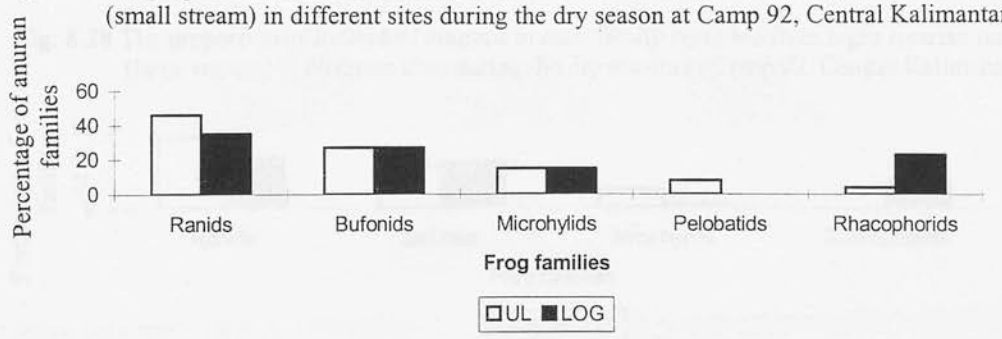


Fig. 8.36 The proportion of individual anurans in each family recorded from night riparian transects (large stream) in different sites during wet season 1 at Camp 92, Central Kalimantan

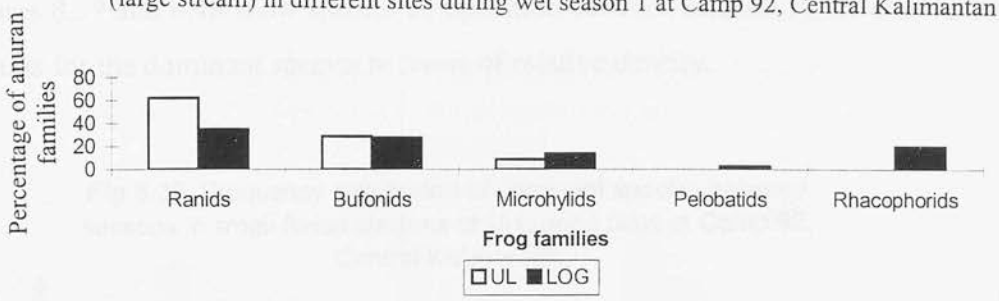


Fig. 8.37 The proportion of individual anurans in each family recorded from night riparian transects (large stream) in different sites during wet season 2 at Camp 92, Central Kalimantan

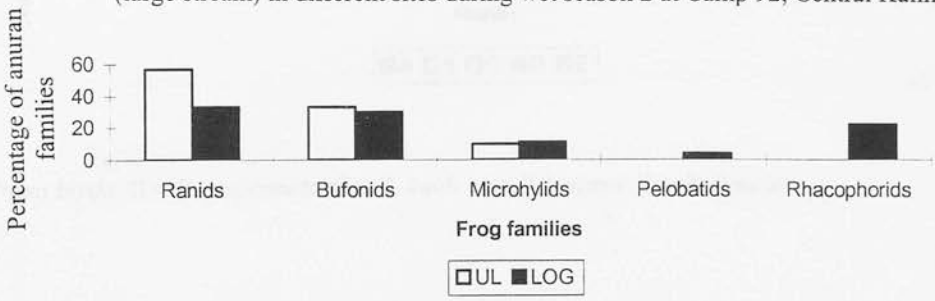
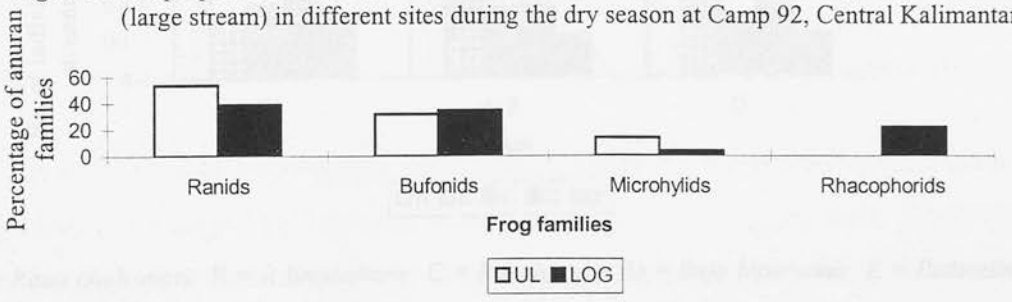
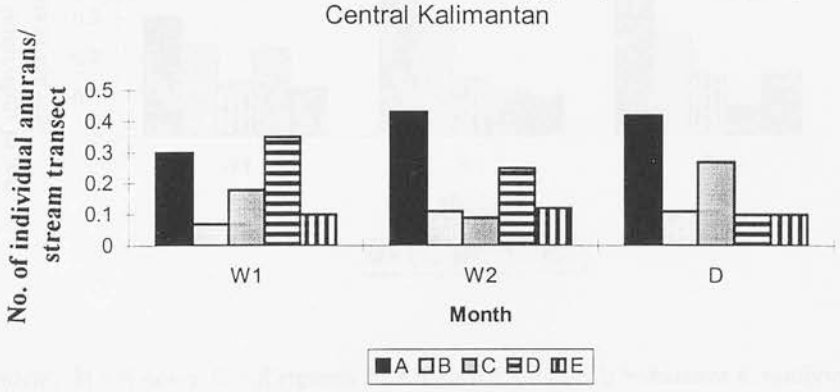


Fig. 8.38 The proportion of individual anurans in each family recorded from night riparian transects (large stream) in different sites during the dry season at Camp 92, Central Kalimantan



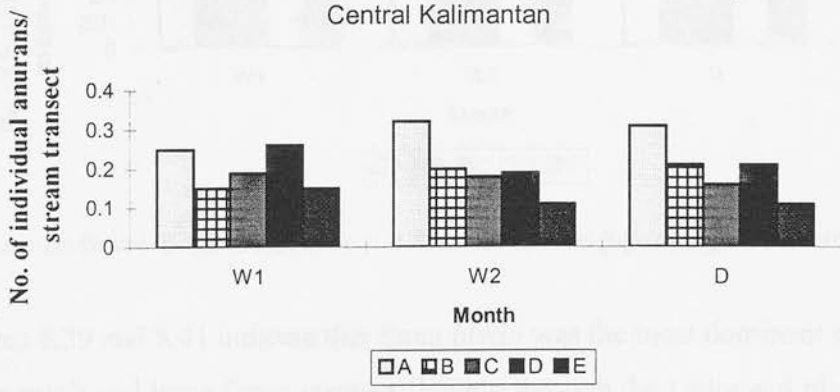
Figures 8.39 and 8.40 show species composition between seasons in the small forest streams for the dominant species in terms of relative density.

Fig 8.39 Frequency distribution of dominant species between seasons in small forest streams of Unlogged plots at Camp 92, Central Kalimantan



A = *Rana blythi* B = *R. chalconota* C = *R. kuhli* D = *R laticeps* E = *R signata*

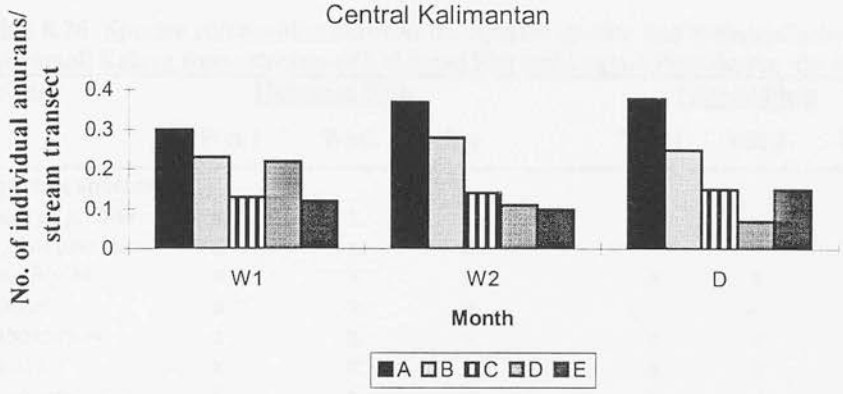
Fig 8.40 Frequency distribution of dominant species between seasons in small forest streams of Logged Plots at Camp 92, Central Kalimantan



A = *Rana chalconota* B = *R limnocharis* C = *R malesiana* D = *Bufo biporcatus* E = *Pedostibes hosei*

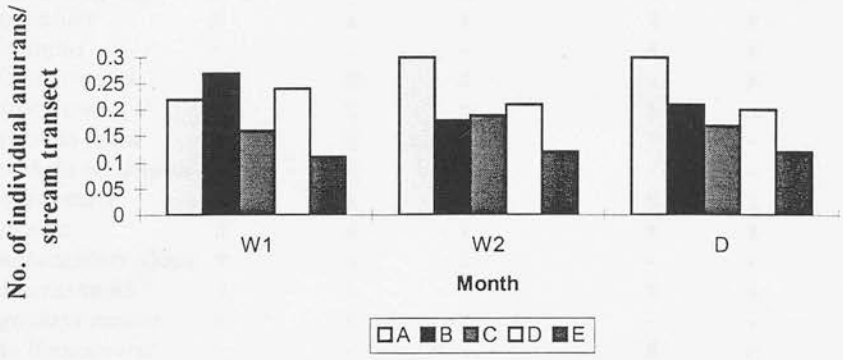
Figures 8.41 and 8.42 show species composition between seasons in the large forest streams for the dominant species in terms of relative density

Fig 8.41 Frequency distribution of dominant species between seasons in the large streams of Unlogged Plots at Camp 92, Central Kalimantan



A = *R blythi* B = *R hosei* C = *R signata* D = *Staurois natator* E = *Ansonia longidigita*

Fig 8.42 Frequency distribution of dominant species between seasons in the large streams of Logged Plots at Camp 92, Central Kalimantan



A = *Rana erythraea* B = *R limnocharis* C = *R malesiana* D = *Bufo divergens* E = *Ansonia leptopus*

Figures 8.39 and 8.41 indicate that *Rana blythi* was the most dominant species found in the small and large forest streams flowing through the Unlogged plots. However, *Rana chalconota* was recorded as the dominant species in the small forest streams of the Logged Plots (Figure 8.40) and *Rana erythraea* was recorded as the dominant species in the large forest streams of the Logged Plots (Figure 8.42).

It is interesting to compare the anuran species composition (see Table 8.26) between riparian species and non-riparian species that were using streams either just for breeding or escape from the forest because their habitats have been disturbed.

Table 8.26. Species composition between the riparian species and terrestrial/arboreal species found in the small Kalang forest streams of Unlogged Plot and Logged Plots during the dry and wet season

Species	Unlogged Plots			Logged Plots		
	Wet 1	Wet2	Dry	Wet 1	Wet 2	Dry
Riparian species						
Amolops jerboa	x	x	x	-	-	-
A. phaeomerus	x	x	x	-	-	-
Rana blythi	x	x	x	x	x	x
R. hosei	x	x	x	-	-	-
R. ibanorum	x	x	x	x	x	x
R. kuhli	x	x	x	x	x	x
R. laticeps	x	x	x	-	-	-
R. malesiana	-	-	-	x	x	x
R. paramacrodon	x	x	x	x	x	x
R. signata	x	x	x	-	-	-
Staurois natator	x	x	x	-	-	-
Bufo asper	x	x	x	x	x	x
B. juxtasper	x	x	x	x	x	x
Terrestrial species						
Ansonia longidigita	x	x	x	x	x	x
A. spinulifer	x	x	x	x	x	x
A. leptopus	-	-	-	x	x	x
Bufo biporcatus	x	x	x	x	x	x
B. divergens	x	x	x	x	x	x
Chaperina fusca	x	x	-	-	-	-
Microhyla berdmorei	x	x	x	-	-	-
M. borneensis	x	x	x	x	x	x
M. ornata	x	x	x	x	x	x
Leptobrachium abotti	x	x	x	-	-	-
K. pleurostigma	x	x	-	x	x	-
Megophrys nasuta	x	x	x	-	-	-
Rana limnocharis	-	-	-	x	x	x
R. erythraea	-	-	-	x	x	x
Arboreal species						
Rana chalconota	x	x	x	x	x	x
Pedostibes hosei	x	x	x	x	x	x
Rhacophorus rufipes	x	x	x	x	x	x
R. appendiculatus	-	-	-	x	x	x
R. leucomystax	-	-	-	x	x	x
R. acurostris	-	-	-	x	x	x
Polypedates macrotis	-	-	-	x	x	x
P. otilophus	-	-	-	x	x	x

Table 8.26 shows that the true riparian species of primary forest such as *Amolops jerboa*, *Rana signata* and *Staurois natator* did not occur in the small streams of

logged stands. In contrast, the disturbed forest species like *Rana limnocharis and R. erythraea* were only present in the small streams of logged stands. The presence of some arboreal species in the logged stands is likely to indicate that their primary habitats have been disturbed. The primary forest species *Chaperina fusca* was observed only in the wet seasons, so it is likely indicate that this species only breeds in the wet season.

Table 8.27. Species composition between the riparian species and terrestrial/arboreal species found in the large Kalang forest streams of Unlogged Plot and Logged Plots during the dry and wet season

Species	Unlogged Plots			Logged Plots		
	Wet 1	Wet2	Dry	Wet 1	Wet 2	Dry
Riparian species						
Amolops jerboa	x	x	x	-	-	-
A. phaeomerus	x	x	x	-	-	-
Rana blythi	x	x	x	x	x	x
R. hosei	x	x	x	-	-	-
R. ibanorum	x	x	x	x	x	x
R. kuhli	x	x	x	x	x	x
R. malesiana	-	-	-	x	x	x
R. paramacrodon	x	x	x	x	x	x
R. signata	x	x	x	-	-	-
Staurois natator	x	x	x	-	-	-
Bufo asper	x	x	x	x	x	x
B. juxtasper	x	x	x	x	x	x
Terrestrial species						
Ansonia longidigita	x	x	x	x	x	x
A. spinulifer	x	x	x	x	x	x
A. leptopus	-	-	-	x	x	x
Bufo biporcatus	x	x	x	x	x	x
B. divergens	x	x	x	x	x	x
Microhyla berdmorei	x	x	x	-	-	-
M. ornata	x	x	x	x	x	x
K. pleurostigma	-	-	-	x	x	-
Rana limnocharis	-	-	-	x	x	x
R. erythraea	-	-	-	x	x	x
Arboreal species						
Rana chalconota	x	x	x	x	x	x
Pedostibes hosei	x	x	x	x	x	x
Rhacophorus rufipes	-	-	-	x	x	x
R. appendiculatus	-	-	-	x	x	x
R. leucomystax	-	-	-	x	x	x
R. acurostris	-	-	-	x	x	x
Polypedates macrotis	-	-	-	x	x	x
P. otilophus	-	-	-	x	x	x

Table 8.27 indicates that in the unlogged plots, species composition in the large streams was similar to that recorded in the small streams. The same results were also

were also recorded in the streams of logged plots. However, several species such as *Chaperina fusca, Rana laticeps, Microhyla borneensis and Megophrys nasuta* that found in the small streams did not occur in the large streams. The species composition in the large streams of the unlogged plots was also different from the logged plots.

8.7. Presence-Absence of Species and Similarity

Rana blythi was encountered most frequently in the small and large forest stream transects of the Unlogged Plots during dry and wet seasons, but in the Logged Plots this species was found on only a few occasions. *Rana erythraea* and *Rana limnocharis* were the most frequently observed in the large forest streams of the Logged Plots but these species were completely absent in the small and large forest streams of the Unlogged Plots. *Rana chalconota* was recorded as a dominant species in the small forest streams of Logged Plots but this species was only found in small numbers in the large forest streams of Logged and Unlogged Plots.

Bufo divergens was one of the dominant species found in the large streams of logged plots, but was only encountered on a few occasions in the small streams of logged plots and all streams in the unlogged plots. The absence of species from the small and large streams of Logged Plots resulted in different patterns of species richness between the Unlogged and Logged Plots.

Table 8.28. Similarities between five dominant species in the Kalang small forest streams of Unlogged Plots between seasons at Camp 92, Central Kalimantan

Coefficient/Species Similarity	AB	AC	AD	AE	BC	BD	BE	CD	CE	DE
Wet 1										
Simple Matching	0.83	0.91	0.91	0.75	0.75	0.58	0.75	0.83	0.75	0.75
Sorenson	0.90	0.95	0.95	0.85	0.84	0.73	0.82	0.90	0.85	0.85
Wet 2										
Simple Matching	0.91	0.75	1.00	0.91	0.83	0.58	1.00	0.75	0.83	0.91
Sorenson	0.95	0.85	1.00	0.95	0.90	0.73	1.00	0.85	0.90	0.95
Dry										
Simple Matching	1.00	1.00	0.83	1.00	1.00	0.66	1.00	0.83	0.83	0.75
Sorenson	1.00	1.00	0.90	1.00	1.00	0.80	1.00	0.90	0.90	0.85

A = *Rana blythi* B = *R chalconota* C = *R kuhli* D = *R laticeps* E = *R signata*

Table 8.28 shows mutual species occurrence for all species A, B, C, D, E is high because all of these species occur together in the surveyed sections except species BD. *Rana chalconota* (species B) usually perch on low vegetation or large boulders while *Rana laticeps* (species D) usually perch either on gravel or sandy substrate.

Table 8.29 Similarities between five dominant species in the Kalang small forest streams of Logged Plots between seasons at Camp 92, Central Kalimantan

Coefficient/Species Similarity	AB	AC	AD	AE	BC	BD	BE	CD	CE	DE
Wet 1										
Simple Matching	0.90	0.75	0.83	0.50	0.66	0.66	0.83	0.91	1.00	1.00
Sorenson	0.95	0.85	0.90	0.66	0.80	0.80	0.88	0.95	1.00	1.00
Wet 2										
Simple Matching	1.00	0.75	1.00	0.50	0.75	1.00	0.75	0.75	0.83	0.75
Sorenson	1.00	0.85	1.00	0.66	0.85	1.00	0.85	0.85	0.88	0.85
Dry										
Simple Matching	1.00	0.83	1.00	0.41	0.83	1.00	0.66	1.00	0.83	0.83
Sorenson	1.00	0.90	1.00	0.66	0.90	1.00	0.80	1.00	0.90	0.90

A = *Rana erythraea* B = *R limnocharis* C = *R malesiana* D = *Bufo biporcatus* E = *Pedostibes hosei*

Table 8.29 shows mutual species occurrence for all species A, B, C, D and E is high for all seasons, except for the species pair AE which was very low because each species has a different type of habitat. *Rana erythraea* (species A) lives on small rocky streams but *Pedostibes hosei* (species E) perches on low vegetation, logs or irregularities of banks.

Table 8.30 Similarities between five dominant species in the Kalang large forest streams of Unlogged Plots between seasons at Camp 92, Central Kalimantan

Coefficient/Species Similarity	AB	AC	AD	AE	BC	BD	BE	CD	CE	DE
Wet 1										
Simple Matching	1.00	0.91	1.00	1.00	0.91	1.00	1.00	0.91	0.91	1.00
Sorenson	1.00	0.95	1.00	1.00	0.95	1.00	1.00	0.95	0.95	1.00
Wet 2										
Simple Matching	1.00	1.00	0.66	0.91	1.00	0.83	0.91	0.75	0.83	0.58
Sorenson	1.00	1.00	0.80	0.95	1.00	0.90	0.95	0.85	0.90	0.73
Dry										
Simple Matching	1.00	0.91	0.83	0.91	0.91	0.83	0.91	0.75	0.83	0.75
Sorenson	1.00	0.95	0.90	0.95	0.95	0.90	0.95	0.85	0.90	0.85

A = *Rana blythi* B = *R hosei* C = *R signata* D = *Staurois natator* E = *Ansonia longidigita*

Table 8.30 shows that mutual species occurrence for all species A, B, C, D and E is high in any season.

Table 8.31 Similarities between five dominant species in the Kalang large forest streams of Logged Plots between seasons at Camp 92, Central Kalimantan

Coefficient/Species Similarity	AB	AC	AD	AE	BC	BD	BE	CD	CE	DE
Wet 1										
Simple Matching	0.91	0.83	0.83	0.45	0.75	0.91	0.50	0.66	0.41	0.58
Sorenson	0.95	0.90	0.90	0.62	0.85	0.95	0.66	0.80	0.66	0.70
Wet 2										
Simple Matching	1.00	0.91	0.83	0.75	0.91	1.00	0.75	0.91	0.66	0.58
Sorenson	1.00	0.95	0.90	0.84	0.95	1.00	0.84	0.95	0.80	0.70
Dry										
Simple Matching	1.00	1.00	0.75	0.91	1.00	0.66	0.91	0.66	0.91	0.75
Sorenson	1.00	1.00	0.80	0.95	1.00	0.80	0.95	0.80	0.95	0.84

A = *Rana erythraea* B = *R limnocharis* C = *R malesiana* D = *Bufo divergens* E = *Ansonia leptopus*

Table 8.31 shows that mutual species occurrence between A, B, C, D, E in general is high for all seasons. However, mutual species occurrence for species pairs AE-BE-DE indicates a very low similarity due to the low number of *Ansonia leptopus* (species E) encountered during wet season 1 and 2.

Table 8.32 Frequency of observations of dominant species of anurans along the length of Kalang large forest streams during wet and dry seasons in the Unlogged Plots at Camp 92, Central Kalimantan

Surveyed Section (m)	Species				
	Rana blythi	*Rana hosei*	*Rana signata*	*Staurois natator*	*Ansonia longidigita*
Wet season					
0-10	3	4	1	3	1
11-20	5	1	2	4	1
21-30	6	3	1	2	1
31-40	5	2	2	2	3
41-50	4	3	3	3	2
51-60	5	2	1	1	2
61-70	5	2	3	4	1
71-80	7	6	4	5	4
81-90	9	7	5	8	4
91-100	5	4	1	3	2
101-110	4	3	2	4	1
111-120	6	4	3	4	2
121-130	5	6	2	3	1
131-140	4	8	2	5	2
141-150	8	7	4	7	4

Table ..... contd

Surveyed Section (m)	Species				
	Rana blythi	*Rana hosei*	*Rana signata*	*Staurois natator*	*Ansonia longidigita*
Dry season					
0-10	3	2	2	0	2
11-20	5	3	0	1	1
21-30	2	3	2	0	2
31-40	3	4	1	0	3
41-50	5	1	0	1	3
51-60	10	4	3	0	1
61-70	8	3	5	2	2
71-80	10	8	6	4	4
81-90	17	10	6	4	5
91-100	8	8	3	0	4
101-110	7	2	2	3	2
111-120	9	4	5	1	3
121-130	5	3	1	0	3
131-140	5	6	1	1	4
141-150	7	8	3	2	3

All dominant species observed during the wet season in the large streams (Table 8.32) were almost equally abundant in all surveyed sections, but they were not equally distributed during the dry season. *Ansonia longidigita* was the only dominant species that was roughly equally abundant in all surveyed sections during both wet and dry seasons.

Table 8.33 Frequency of observations of dominant species of anurans along the length of Kalang small forest streams during wet and dry seasons in the Logged Plots at Camp 92, Central Kalimantan

Surveyed Section (m)	Species				
	Rana chalconota	*R limnocharis*	*R malesiana*	*Bufo biporcatus*	*Pedostibes hosei*
Wet season					
0-10	1	0	1	1	0
11-20	1	0	0	1	0
21-30	2	1	1	2	0
31-40	0	0	1	3	1
41-50	2	0	1	1	1
51-60	0	2	2	2	0
61-70	1	1	0	1	1
71-80	3	0	0	1	2
81-90	0	1	3	2	1
91-100	2	1	0	1	2
101-110	1	3	1	0	0
111-120	2	0	0	3	0
121-130	1	1	2	1	2
131-140	1	2	1	0	1
141-150	3	0	2	2	1

Table.....contd.

Surveyed Section (m)	Species				
	Rana chalconota	*R limnocharis*	*R malesiana*	*Bufo biporcatus*	*Pedostibes hosei*
Dry Season					
0-10	2	2	1	2	1
11-20	1	1	1	1	1
21-30	4	1	2	3	0
31-40	1	3	1	2	0
41-50	3	2	1	2	1
51-60	5	1	1	1	1
61-70	4	4	2	4	0
71-80	6	4	3	3	1
81-90	5	3	3	4	2
91-100	2	1	2	1	1
101-110	2	2	1	1	2
111-120	5	1	2	2	1
121-130	4	3	1	3	2
131-140	2	2	1	1	1
141-150	2	3	3	2	2

Table 8.33 shows all dominant species in the small forest streams flowing through the Logged Plots during the wet season were not frequently seen and they all were not equally abundant in all surveyed sections. In contrast, all species appeared in all sections of the surveyed streams during the dry season except *Pedostibes hosei.*

Table 8.34 Frequency of observations of dominant species of anurans along length of Kalang large forest streams during wet and dry seasons in the Logged Plots at Camp 92, Central Kalimantan

Surveyed Section (m)	Species				
	Rana erythraea	*R limnoharis*	*R malesiana*	*Bufo divergens*	*Ansonia leptopus*
Wet season					
0-10	2	1	2	1	1
11-20	6	2	3	2	0
21-30	7	4	3	3	1
31-40	8	4	2	2	2
41-50	6	4	5	4	2
51-60	2	1	1	2	1
61-70	3	1	1	2	1
71-80	3	2	1	2	2
81-90	5	2	4	4	3
91-100	1	2	1	4	2
101-110	2	2	1	1	1
111-120	1	1	1	2	1
121-130	1	3	4	5	2
131-140	3	1	2	1	0
141-150	3	3	4	2	2

Table ... contd.

Surveyed Section (m)	Species				
	Rana erythraea	*R limnoharis*	*R malesiana*	*Bufo divergens*	*Ansonia leptopus*
Dry Season					
0-10	3	0	1	1	0
11-20	6	4	3	4	3
21-30	7	6	6	5	3
31-40	6	5	4	6	4
41-50	7	6	5	5	4
51-60	2	1	0	1	0
61-70	0	0	1	2	1
71-80	6	3	1	1	1
81-90	8	5	4	4	3
91-100	3	0	1	1	0
101-110	0	2	0	2	1
111-120	2	1	1	1	1
121-130	5	4	3	3	2
131-140	0	0	1	1	0
141-15	7	6	5	4	3

Table 8.34 shows that each of these dominant species during the wet season appeared in all sections of the large forest streams flowing through the Logged Plots, though usually with greater frequency in one portion. *Rana erythraea*, for example, was much more abundant in the lower half of the surveyed section. In contrast, not all of these species appeared in all sections of the surveyed streams during the dry season. *Bufo divergens* was the only dominant species which appeared in all sections of the stream surveyed.

With the exception of *Pedostibes hosei*, the dominant species are only rarely (<3%) found more than 30 ft (9m) from a stream bed. Within this 9m-zone, all the dominant species had slightly differing distributions (Tables 8.31 and 8.32). These distributions overlapped extensively. *Rana signata, R. chalconota, R hosei, Pedostibes hosei and Ansonia longidigita* were often found perched on vegetation within 6ft (1.8m) of the ground though many were observed on the rocks or soil of the banks. The rest of the dominant species (Tables 8.33 and 8.34) were always seen on the ground.

8.8. Species Distribution

Table 8.35 Distribution of anurans in the surveyed section of Unlogged plots with respect to distance from beds of Kalang small and large forest streams at Camp 92, Central Kalimantan

Species	% observed up to 1.5m from stream bed	Maximum distance (m) from stream bed
Kalang Small Stream		
Rana blythi	40	5
R chalconota	80	3
R kuhli	50	6
R laticeps	85	3
R signata	95	3
Kalang Large Stream		
Rana blythi	30	9
R hosei	55	8
R signata	85	5
Staurois natator	75	6
Ansonia longidigita	60	9

Table 8.36. Distribution of anurans in the surveyed section of Logged plots with respect to distance from beds of Kalang small and large forest streams at Camp 92, Central Kalimantan

Species	% observed up to 1.5m from stream bed	Maximum distance (m) from stream bed
Kalang Small Stream		
Rana chalconota	40	6
R limnocharis	60	8
R malesiana	70	4
Bufo biporcatus	85	5
Pedostibes hosei	65	3
Kalang Large Stream		
Rana erythraea	50	10
R limnocharis	45	12
R malesiana	55	6
Bufo divergens	70	6
Ansonia leptopus	60	8

The degree of clustering can be measured by the maximum percentage of observations in one 20-ft (6m) interval (Inger 1969). For *Rana blythi* the maximum percentage on each of the small and large streams varied from 4 to 7%, *R laticeps* (4-6%), *R kuhli* (6-8%), *R malesiana* (4-8%) and *Bufo asper* (4-8%) had equivalent, low percentages. So, they only had small clusters. *Rana hosei* (8-14%) and

Pedostibes hosei (10-14%) had larger maximum percentages. *Pedostibes hosei* shows the strongest tendency towards clustering. The other two dominant species, *Rana signata* and *R chalconota*, formed clusters that were obvious in the field. The maximum percentage of observations in one 20-ft (6m) interval varied from 15 to 28% for *R signata* and from 18 to 27% for *R chalconota.*

What characteristics of these various species of anurans are associated with their varying tendencies to form clusters? Size might be expected to figure in the spatial distribution of individuals within a community. Presumably, within a given trophic level a larger species will require more food, hence a larger feeding range, and need more space between individuals than would be true of smaller species. In the present assemblage of anurans, the weakly clustered species are indeed larger though the group includes one small to medium species and one large species in the more strongly clustered group (Table 8.37).

Table 8.37 Association of clumping tendency with other characteristics of dominant anurans from Kalang rain forest streams in Camp 92, Central Kalimantan

Species	Clumping tendency	Males call	Sex ratio (males to total adults)	Snout-vent of adults (mm) Males	Females
Rana signata	Strong	+	0.65	33-47	48-68
R chalconota	Strong	+	0.73	33-44	47-59
Pedostibes hosei	Moderate	+	0.90	53-80	88-105
Rana hosei	Weak	+	0.25	50-60	85-100
R malesiana	Weak	0	0.40	80-130	75-128
R kuhli	Weak	0	0.49	80-115	80-104
R blythi	Weak	0	0.45	90-170	85-125
Bufo divergens	Weak	+	0.30	70-98	95-120
Ansonia longidigita	Weak	+	0.42	35-49	45-64

The group with only weak clustering tendencies includes voiceless species as well as those that call, but all species that have moderate to strong clumping tendencies do call (Table 8.37).

8.9. Tadpole Habitats and Breeding

Kalang forest streams in Camp 92 furnish anuran larvae with a variety of habitats, which I can group in a graded series according to current. At the swift end of the scale are stretches of strong rapids and small waterfalls where the bottom is formed of large boulders and bed rock. As the gradient drops and the current is reduced, the bottom changes to garvel and coarse sand. In places, especially where a bank is undercut, root masses project into the streams and form pockets of still slower current. Gravel bars cut off pools usually have a thick layer of dead leaves and a thin layer of silt. These side pools often have no surface connection with the main current after a few days without rain. Falling water levels after heavy rains usually leave two kinds of pools off to the side of the stream bed and 0.3-0.9 m above the level of the bed. These isolated pools may be shallow depressions in the flood plain or pot-holes in out-cropping rock.

Within the short sections of the streams I worked, these parts of the drainage system are not sorted with respect to the longitudinal axis of the stream. They have the relation described only at normal water levels. By 'normal' I mean levels associated with clear water. Immediately after rain the water rises and becomes turbid. A light rain (c. 1.3cm) is sufficient to effect junction between all parts within the stream bed, while a heavier rain (2.5 cm) usually raises the water level enough to engulf the pools in the flood plain.

The species tadpoles are not unigormly distributed in these habitats, though the quiet side pools with beds of dead leaves provide most species with their main habitat. In many instances, several species of larvae were collected in one location and in some cases the joint occurrences involved closely related species. Transforming young Leptobrachium, Megophrys nasuta, Rana blythi were collected together at the edge of a side pool. In December 1994 I observed two pairs of arboreal frogs perched on low vegetation at a height about 1m near the streamside of small Kalang streams. In January 1995, I also observed a pair of Leptobracium abotti perched near the stream bank of small Kalang river.

8.10. Discussion

The abundance of anurans observed per stream transect in the small and large streams showed a variation (Figures 8.3 - 8.8) with season. The only trend that emerges from this variation may obscure seasonal behaviour of each species. If a species was more active at one time of the month than at another, temporal variation in number of anuran species may occur. Correlation of these fluctuations with rainfall is weak, as it is with humidity or temperature. Consequently a systematic interpretation of these fluctuations can be achieved only through deeper investigation and long term monitoring to get more reliable evidence. The overall picture for the five dominant species observed in the stream transect is that there was little evidence that can be called seasonal or periodic changes in numbers.

There was a significant difference in abundance of anurans per stream transect between streams flowing through the Unlogged and Logged Plots (Table 8.4 and 8.5). Species composition in the streams of Unlogged stands also differed from the Logged stands. One possible explanation may be the difference in physical characteristics of the streams between Unlogged and Logged stands. One major physical characteristic that differs between streams in these two different stands was that Logged Plots had generally finer sediments (Figure 8.1 and 8.2). From field observations, *Rana blythi* was the dominant species observed in the streams of Unlogged Plots and mostly found in the clear rocky streams. Whilst *Rana erythraea* was the dominant species recorded in the streams of selectively Logged Plots and were observed mostly in the turbid water with silty sandy bottom.

Canopy cover, air humidity and air temperature in the primary forest streams were also different from the logged stand streams. Linear regression analysis indicated that there was a strong relationship between humidity and temperature and the abundance of anurans. The mixture of these two microclimatic components affects the distribution and activity pattern of the anurans (Trueb and Duelmann 1994). Canopy cover also significantly affected the abundance of anurans (Table 8.3). The linear regression analysis showed that the relationship between canopy cover and

abundance of anurans is stronger in the dry season than in the wet season. This suggests that the function of canopy cover during the dry season may be more important than in the wet season to maintain the humidity that is vital for the anurans in reducing the loss of water because of dry conditions. Table 8.11 also showed that percent canopy cover had a significant effect on abundance of anurans. Moreover there was a significant effect of canopy cover on species richness during the dry season in the small and large streams (Table 8.20 and 8.21). This indicated that the role of canopy cover during the dry season is more important in maintaining the humidity and reducing the loss of water that is important for anurans.

Species overlap depended upon habitat preferences in the surveyed streams. For example, the similarity coefficient between *Rana erythraea* and *Pedostibes hosei* was very low because they did not actually occur together in the same place. *Rana erythraea* is usually found on a silt sandy bottom whilst *Pedostibes hosei* was most likely to perch on rocky substrate. High similarity coefficients for *Rana erythraea* and *Rana limnocharis* showed that they both occupied the same habitat (Table 8.29).

A species may appear to be clumped in the surveyed section because of repeated observations of a few individuals that did not move much. Each of the small and large streams varied along its length. There were bends, gravel bars, riffles, pools, steep banks, etc. at odd intervals. If a species of anuran is better adapted to any particular combination of physical circumstances, there would have been favourable and unfavourable places for it along a stream. A concentration would be expected or small local population to be centered at every favourable situation with individuals scattered about the best spot within each centre. In addition, there was a tendency for greater clumping in the dry season. It is likely that anurans may react unfavourably to the alteration of hydroperiods, and because the humidity becomes patchy, they tend to be more concentrated in humid places where they are more likely to survive.

CHAPTER IX

DISCUSSION AND CONCLUSION

Three different sampling techniques were used in assessing the impact of logging on the amphibian community in unlogged and logged areas of Kalimantan rainforest. The first technique, drift fences with pitfall traps was used to sample amphibian assemblages on the forest floor. This technique was the most passive sampling design. The second technique was searched quadrat (area-constrained searches), an active sampling method, while the third technique, night riparian transects was used to sample the riparian anuran species. The different sampling techniques applied in the experimental plots, different type of habitats and time of samplings between pitfall traps and searched quadrats with night riparian transects meant that the results obtained with the pitfall trap and searched quadrat techniques were not comparable to the night riparian transect technique.

9.1. Pitfall Traps and Searched Quadrat

Abundance of anurans

The results of pitfall traps and drift fences showed that there were significant differences in anuran abundances between the forest control plots (unlogged plots) and the 2-year-old logged plots. Although the survey sampling techniques between pitfall traps and searched quadrats were different, they obtained the same results. Thus, the results reported in Chapter VII strongly support those presented in Chapter VI. The study results are in line with the results of most studies examining the effects of clear-cuts on amphibians, which reported higher abundances in forest control plots than in logged plots with both active sampling techniques using area- and time-constrained searches (Bury 1983; Pough *et al.* 1987; Ash 1988; Corn and Bury 1989; Dodd 1991; Petranka *et al.* 1993, 1994; Dupuis *et al.* 1995), and most passive sampling designs using pitfalls or cover boards (Blymyer and McGinnes 1977; Enge and Marion 1986; Bury and Corn 1988; Raphael 1988; Corn and Bury 1991; De Graaf and Yamasaki 1992; Foley 1994).

Differences in abundance may reflect differences amongst other variables such as amphibian species, forest type, elevation, microhabitat and climate, and age discrepancies between control stands and logged stands (Pais *et al.* 1988, Phelps and Lancia 1995 and Clawson *et al.* 1996). This study's results indicate that there was a significant difference in vegetation canopy covers and the microclimate components, in particular humidity and temperature, between unlogged or closed canopy logged plots and open canopy logged plots.

The reduction of canopy cover following logging has important effects on the physical environment of tropical rain forests. Logging creates artificial canopy gaps with a variety of sizes and shapes that vary with the method and intensity of logging (Canon *et al.* 1994). This ranges from areas of relatively undisturbed forest under closed canopy to large canopy gaps or open canopy areas of logged forests (Canham *et al.* 1990).

The open canopy areas of logged plots in the study sites have large canopy gaps, creating more extreme conditions of microclimate with lower humidity and high temperature (as compared to unlogged plots). This observation is in line with Thiollay (1992) who mentioned that the microclimate had obviously changed from typically dark, humid, cool, and almost wind-free undergrowth in the unlogged plots to increased light levels, mid-day temperature, and wind turbulence and lowered humidity in the open canopy logged plots.

Canopy structure influences most physical processes of importance in the forest ecosystem. The most obvious of which is to increase the amount of radiation penetrating the canopy to the forest floor. The change in the radiation balance of forest affects many biological and physiological processes, notably sub-canopy microclimate. The energy balance of the forest is altered because of increased penetration of radiation into the lower levels of the canopy and to the forest floor. This in turn affects other aspects of the physical environment including air and soil temperature, humidity and the water balance of the forest (Raich 1989). The change in the energy balance of the forest in large gaps following logging leads to increased

temperature and decreased relative humidity (Ashton 1992, Brown 1993). In this study, the average humidity decreased from 94.5% (wet season) and 91% (dry season) in the unlogged and closed canopy-logged plots to only around 89% (wet season) and 82% (dry season) in the open canopy logged plots. Whilst the average temperature increased from 24.5°C in the unlogged and closed canopy-logged plots to up to 27°C in the open canopy-logged plots in the dry season as described in Chapter V. The significant differences in humidity and temperature were due to the presence of large canopy gaps in the open canopy logged plots.

Humidity and temperature are very important for anurans because these factors affect the timing and intensity of their activities (Trueb and Duellman 1994). The mixture of temperature and humidity determines the rate of water loss from the skin of amphibians; therefore, the amount of moisture in the air significantly affects their distribution and activity pattern. Moreover, the existence of a pronounced dry season will exacerbate the effects of decreased humidity, in more open areas such as in the open canopy logged plots. Linear regression also showed that the abundance of anurans was related to humidity and temperature (Chapter VI and VII). This is likely to affect anurans which so much depend on the high humidity and cool temperature for their survival.

Most of the forest-dwelling species either captured in the pitfall traps or observed in the searched quadrats were species that are highly dependent on water. These species may react unfavourably to increased insolation and decreased humidity in the altered habitat of open canopy-logged plots. Intensive logging which creates large canopy gaps, usually decreases plant litter and thereby affects the anurans that depend on its presence for refugia (Heyer *et al.* 1994). This suggestion is in line with the study results that forest litter cover was significantly related to abundance of anurans in all sites (Chapter VI and VII).

Several activities of timber operations may affect the amphibian's forest floor habitat. Logging activity which creates large canopy gaps usually decreases plant litter, because intensive logging practices disrupted forest litter and reduced inputs from the leaves that are falling to the forest floor. This will certainly affect the activity of forest

litter anurans that are highly dependent on the presence of forest litter cover for their main refugia (Pough *et al.* 1987; Bonin 1991). In addition, drying of leaf litter and humus, due to canopy removal, limits amphibian movements and the ability to forage as well as reducing the abundance of anurans (Pough *et al.* 1987, Dodd 1991, Dupuis *et al.* 1995). If the canopy in a logged site is eliminated and the forest floor becomes relatively dry in the large gaps of open canopy areas (as compared to a forested site or closed canopy of logged forest), then amphibians would be expected to be able to forage less often and reduce their activities. The process of forest ecosystem recovery takes several years, therefore the decrease in numbers of anurans in such areas of open canopy logged plots would be expected to last until the relative humidity increased due to canopy regeneration. In addition, the reduction of canopy cover in the open canopy logged plots may indirectly affect local distribution of anurans by altering the moisture and temperature which are important for their survival. Consequently, a relationship between forest-dependent anuran distribution and forest litter cover might be expected. The results of this study are in line with the results from several studies (Heatwole 1962, De Graaf and Rudis 1990; Bonin 1991) that showed forest litter had an important relationship with amphibian abundance. Their results suggest that intensive forest harvesting practices that minimize soil compaction and litter disruption might shorten the length of recovery time for amphibian species associated with this microhabitat. After intensive logging, for example, reduced inputs combined with increased rates of decomposition lead to a decline in forest litter. The time required for litter recovery to predisturbance levels can be quite long in some forest types, e.g., up to 50-80 years in northern hardwood forests (Likens et al. 1978; Hughes and Fahey 1994).

The number of anurans in the unlogged plots was almost the same as in the closed canopy logged plots because the forest conditions between the two forests were very similar in terms of the closed canopy cover and percentage plant cover (Chapter V). In addition, the results of a study on the impact of selective logging on growing conditions and on vegetation damage in Central Kalimantan show that natural gaps in unlogged forest were about 5% and canopy gaps in the closed canopy logged forest were about 8%. In contrast, canopy gaps in the open canopy logged forest were about 40% (Clearwater, pers com.). Therefore, there were no significant differences in

humidity, temperature and plant cover between the unlogged and closed canopy logged plots. As a consequence, the abundance values in these two similar forests were not significantly different (Table 6.2 and 7.1). This may suggest that low intensity of logging which creates only small gaps and does not really affect plant/litter cover may not reduce the number of amphibians. If all logging was done as in the closed canopy logged plots, then anurans (and a lot of other animals) would be safe. In contrast, the number of anurans in the open canopy logged plots was much lower than in the unlogged and closed canopy logged plots. This indicates that logging decreases the number of terrestrial amphibians, probably by reducing the availability of moist microhabitats and limiting foraging and reproductive opportunities. This suggests a narrower window of activity for anurans in the logged stands, which may in turn reflect poorer habitat quality. Thus, it could be important to maintain some old-growth tracts as a source of species, particularly if the logged stands represent sink habitats for sustainable wildlife management and may also represent a stronghold for anuran survival in the logged stands. The greater abundance of anurans in unlogged plots demonstrates the importance of unlogged forests to this group of vertebrates.

In general terms, most reduction in animal abundance after logging is seen among vertebrate species specialized to exploit resources that are suddenly limited (eg. primates and birds) (Leighton and Leighton 1983, Karr 1986, Johns 1988), or species (in particular amphibians) restricted by the microclimate changes observed in the logged areas (Wake and Blaustein 1990). Species that occupy narrow niche widths have a statistically higher probability of elimination by logging than species with wider ones. This is particularly marked among groups of amphibians which did not move much and had limited dispersal ability. For example, *Leptobrachium abotti* as a primary forest-dependent species did not occur in the open canopy logged plots.

Plant cover such as ground cover, herb cover and canopy cover had no significant effect on abundance of anurans in the experimental plots. As mentioned earlier, most of the anurans either captured in the pitfalls or observed in the searched quadrats were forest-dwelling species. These anurans are typically ground-dwelling species that depend on litter cover for their main refugia, therefore, plant cover had no direct

influence on their abundance. This observation is in accord with other work. For example, Bury (1983) stated that plant cover apparently was not responsible for the differences in terrestrial amphibians occurring in native or harvested redwood (*Sequoia sempervirens*) forest in Redwoods National Park, western California. Rather, microclimatic and plant litter changes that occur after logging apparently resulted in unsuitable conditions for many amphibians. Woinarski and Gambold (1987) concluded that the distribution of forest litter anurans is associated mostly with substrate variables (notably rock cover and litter cover), moisture and water availability, rather than measured vegetation attributes.

Species richness and composition

The results from pitfall traps and searched quadrats indicate that mean species richness varied slightly between sites in different seasons but there were no significant differences either between sites or between seasons. Pim (1979) stressed that a number of disturbances caused by intensive selective logging affecting complex forest ecosystems are more likely to cause reduction in numbers (abundances) rather than species richness. This was possibly because selective logging does not reduce drastically the diversity of habitat in comparison to clear felling which in turn may affect species richness. In Sabah, Malaysia, selective logging had no significant effect on the species composition of ground cover layers and the relative proportions of different tree species, and only a small proportion of pioneer species occurred after disturbance (Johns 1989). Enge and Marion (1976) stated that there was a positive correlation between amphibian species richness and habitat diversity after clear felling in North Florida Flatwood forest. The clear felling had drastically changed the species composition of gound vegetation which created a more diverse habitat at ground level and increased species richness of the amphibians. The number of amphibian species in a certain habitat is generally affected by the plant diversity of that habitat (Blaustein 1994). Since the effects of selective logging in this study had not significantly reduced the plant diversity of the habitat, it is understandable that logging did not affect the species richness (Heyer *et al.* 1994; Blaustein 1994; Gascon 1991; Inger and Stuebing 1989). However, there was a significant effect of litter cover on species richness during the dry season. The patchiness of humidity and high temperature in the dry season caused the anurans to find humid and cool places for

their refugia. Litter cover is likely to provide constant cover from adverse weather and also an important source of moisture throughout the season (Feder 1983; Corn and Bury 1990), therefore more species were found in the litter cover during the dry season. This suggests that logging practices should be done more carefully during the dry season and minimize forest litter disruption as much as possible, because they are important as refugia for forest litter-dependent species.

Species composition was quite different between the unlogged and open-canopy logged plots. Certain species appear to be characteristic of particular forest types. For example, *Leptobrachium abotti* found only in the Unlogged and Closed Canopy Logged Plots. In contrast, *Rana limnocharis* found only in the Open Canopy Logged Plots. *Rana limnocharis* is a common pioneer in logged forest and frequently found in disturbed habitats associated with human activity such as roadsides and logging roads (Inger and Stuebing 1992). In addition, this ranid is a common species, inhabiting various environments and is more adaptable to extreme conditions with broad tolerance to temperature and moisture fluctuations (Duellman and Trueb 1994); it even has higher operating temperatures (Stebin and Cohen 1995). Perhaps this is related to an ability of this frog to store and reabsorb large quantities of water in the bladder (up to 20-30% of its body mass: Zug 1993), and this affords it a greater tolerance of warmer and dryer climates such as in the open canopy logged plots. Therefore, it is not surprising that the two dominant species *Rana limnocharis* and *Bufo biporcatus* were more frequently sampled in pitfalls and searched quadrats in the logged plots. Moreover, most of the anurans either captured or observed in the altered open-canopy logged plots were species that do not seem to depend so much upon the microclimatic conditions of the forest floor. Many of these are opportunistic breeders, breeding wherever habitat conditions are favourable for the development of tadpoles (Nawangsari 1987). It is argued that, being opportunistic species, they are less subject to climate variations and are active throughout the year (Andreone 1994).

Unlike the disturbed forest species, *Leptobrachium abotti* and *Kalophrynus pleurostigma* are two species which appear in primary rain forest that are active on the ground, which have low mobility and are easy to trap (Inger and Stuebing 1992). Several studies also reported the capture of large numbers of the microhylid (narrow-

mouthed toad *Gasthrophyne carolinensis*) using pitfalls and drift fences because of their low mobility and high capture rates (Campbell and Christman 1982, Enge and Marion 1986, Mengak and Guynn 1987).

Inger and Stuebing (1989) proposed that pelobatids like *Leptobrachium abotti* could serve as a reliable indicator of vital limits and optimum climate, because they cannot survive in disturbed habitat. My study results agreed with Inger and Stuebing, because no pelobatids were captured in the open-canopy logged plots, indicating that this group of species may be considered as primary forest-dwellers and they cannot survive in the harsh environments with low humidity and extreme temperature. One possible reason for this is that the smooth skin layer of these anurans renders them very sensitive to temperature and humidity. This is in contrast to bufonids which have very rough skin that can adapt to harsh climate conditions. Although the integument is the structural and functional interface between the organism and its environment, the morphological and functional complexity of amphibians' skin is incompletely understood and few studies have investigated the effect of microclimates on its functional complexity (Lindemann and Voute; Whitear 1977).

9.2. Night Riparian Transects

Niche characterization

It is a commonplace observation that species living in the same area differ ecologically. Examples can be drawn from almost any group of animals. These results are expected, given the morphological and physiological differences between species. The niches of the dominant species of anurans along these Kalang forest streams overlap to varying degrees. From the observations recorded, it may be suggested, these species may be divided into four general ecological types and overlap of niche between these groups is extensive.

1. Large (Snout vent length more than 80mm), strictly terrestrial, weakly clustered, riparian anurans: eg. *Bufo divergens*, *Rana blythi*, *Rana malesiana* and *Rana kuhli*.
2. Small (Snout vent length less than 40mm) to large, partially arboreal, weakly clustered, riparian anurans: eg. *Rana hosei* and *Ansonia longidigita*. These differ in size.

3. Small, partially arboreal, strongly clustered, riparian anurans: eg. *Rana signata* and *Rana chalconota*. *R. chalconota* has stronger arboreal tendencies than *R. signata.*
4. Large, mainly arboreal, strongly clustered anurans; not strictly riparian: eg. *Pedostibes hosei.*

Overlap of niches between these ecological types is indicated by the general similarity of diets, the lack of complete spatial separation of adults, and the utilization of quiet pools by tadpoles of all anuran species, except *Rana hosei* and *Pedostibes hosei* (Inger 1969).

Two alternative hypotheses may be proposed, though not completely exclusive, to explain the distribution of species in these four ecological types. The first, suggested by various authors (e.g. Da Cunha *et al.* 1951; Kohn 1959), is that natural selection tends to reduce the amount of overlap in niches so that ecological isolation of sympatric species is favoured. The second is that the phylogeny of any given species is more important in determining that species' ecological role than is relatively recent selection for ecological separation.

If the distribution of species of a particular taxonomic group be considered in the four ecological types, they may be (a) distributed at random with respect to one another, (b) distributed in separate ecological types, or (c) grouped in a single type. The implications of each arrangement are: (a) little evidence either for selection for ecological isolation or for phylogenetic determination of ecological behaviour; (b) little phylogenetic determination, but strong selection for ecological isolation; (c) little selection for isolation, but strong phylogenetic determination. Results obtained here suggest that there are two phylogenetic stocks containing more than one species: the group of *Rana* with three species and the *Hylarana* group with two. Each stock occurs in only one of the ecological types; each has distribution (c). The probability of this distribution for both groups on the basis of chance alone is very small. On these grounds, therefore, the hypothesis of phylogenetic determination seems the most likely explanation of the observed distributions.

This analysis can be extended to the other less common species such as strictly riparian and arboreal species and species that use streams only for breeding purposes. Considering only those genera with more than one species, a generally similar pattern may be found. The strictly arboreal species are those in the genus *Rhacophorus*. The two genera, *Leptobrachium* and *Megophrys* live on the forest floor and come to streams only for breeding purposes. Only one species of *Staurois* (strictly riparian) failed to support the conclusion reached for the dominant species, only 25% of fifty-eight *S. natator* were caught on the ground; the rest were perched on vegetation.

Stream communities

What connection, if any, is there between the characterization of niches and the comparison of communities on small and large streams? Perhaps the discussion should be limited to the larger streams since they seem to be generally similar in terms of their community structure and since the characterization of niches in this study deals mainly with the dominant species on these streams.

The communities are not identical with one another, differing in the relative frequencies of particular species. Yet the relative constancy of the frequencies of ecological types from stream to stream and the similarities suggest that a given species can be 'plugged into' the system or removed from it without seriously changing the system. Does this mean that species can be added or subtracted indefinitely without affecting community structure? Obviously they cannot be subtracted indefinitely; the community would disappear. But some subtraction and addition certainly seems possible. *Rana macrodon*, for example, was observed only ten times on Kalang large streams during the entire surveyed transects; in effect it was virtually absent. If the ecological survey of the anuran community on Kalang large streams had been analysed first and followed by that in Kalang small streams, a new and important species, *R macrodon* would have been added to the species list, without appreciably altering the structure of the community. Similar results were gained in earlier experimental plots investigated by Djoko (1989) in North Kalimantan forest streams.

If additions of species are possible within this ecological framework, then competitive exclusion may not be a general phenomenon among anurans living along these and similar rain forest streams. In fact, the contrast between the communities of the large and small streams suggests that at least in this rain forest area, the presence and absence of a riparian species is as likely to be determined by the physical characteristics of a stream as by the presence of ecologically related species. That is, the physical characteristics determine the success of particular ecological types, each of which may include a number of species.

Abundance of anurans and species richness

There was a significant difference in abundance of anurans surveyed per stream transect between small streams flowing through unlogged and 2-year-old logged forests as well as the large streams in the study sites. One major physical characteristic that differs between streams in the Unlogged and Logged Plots was that Logged Plots had generally finer sediments. Canopy cover, humidity and temperature in the Unlogged Plots were also different from the Logged Plots.

The action of sediment in streams is complex, depending on several factors including geomorphology, gradient, logging practices, and amount of road building (Burns 1972; Swanson and Lienkaemper 1978; Adam and Beschta 1980; Murphy *et al.* 1981; Hawkins *et al.* 1983; Reid and Dunne 1984; Duncan and Ward 1985). Logging and associated road construction often result in increased sediment loads in forest streams (Burns 1972, Beschta 1978 and Rice *et al.* 1979). Most of the increased fine sediment observed in the logged sites was derived from logging practices, because there were apparently no road crossings upstream. In addition, the road on steeper slopes near the streams in the logged plots was a more recent construction but is already eroding badly and no special preventions were carried out in its construction. In many places the eroded material from the road had reached the stream system, and was causing high silt deposits on stream beds and creating cloudy water (*pers. obs*).

Inger (1979) stated that the high sedimentation rates in the logged areas affected the development of tadpoles and reduced the abundance of anurans. Murphy *et al.* (1981)

also observed that increased sedimentation apparently caused habitat destruction through filling of substrate crevices and reduced the riparian species populations.

Increased sedimentation in the Logged Plots is probably more important in determining the presence and size of *Rana erythraea* and *R limnocharis*. Inger and Stuebing (1989) reported that abundance of these two species was related to the number of substrate crevices, and crevices in the low-gradient streams in the logged sites were filled by fine sediment. These two species also have broad tolerance to the extreme changes of temperature and even they had a wider range of habitats (Boeadi 1987).

As mentioned previously, canopy cover had a significant effect on abundance of anurans in small and large streams (see Table 8.10 and 8.11). The reduction of canopy cover in the logged stands may indirectly affect the local distribution of anurans by altering the moisture and temperature of cover objects (Heatwole 1962, Pough *et al.* 1987). For example, shaded forest streams can be three-and-a-half to five times more moist and up to five times cooler than exposed ones (Heatwole 1962). Available forest canopy cover for anurans therefore is important not only to maintain the humidity but also reduce the water loss. Therefore, there is a significant effect of canopy cover on abundance of anurans. Moreover, there was a very significant effect of canopy cover on species richness (Table 8.20 and 8.21) in the dry season. This indicates that canopy cover is as important in affecting the abundance and species richness of riparian anurans, especially during the dry season when humidity is very limited.

Shade, like canopy cover, is urgently needed for riparian species, because beside streams there is almost no litter cover which is important to provide moist and cool habitat for their survival. Rudolph and Dickson (1990) reported that only 13.5% of total captures were made in narrow (0-25m) strips beside streams in the southwest Pacific forest, compared to 50% of total captures made in wide strips located 50-100m from the stream. The differences in abundance of anurans were mainly because there was no litter cover immediately beside the streams compared to further away where there was thick leaf litter. Therefore, it is vital to leave streams with full canopies which can function as corridors between undisturbed areas of forest. The streams

provide important breeding habitat for riparian as well as non-riparian species. The results in Chapter VIII (table 8.26 and 8.27) indicate that certain species like the forest-dependent species *Chaperina fusca*, *Leptobrachium abotti* and *Bufo biporcatus* used the streams for breeding. I have also observed two pairs of arboreal frogs *Rhacophorus rufipes* perched on low vegetation at a height of about 1 m near of small Kalang streams. In addition, I also found some tadpoles of *Leptobrachium abotti* and *Megophrys nasuta* in the pools of large Kalang streams (Taufik, field obs.). Several arboreal species such as *Rhacophorus appendiculatus*, *R. leucomystax*, *R. acurostris* and *Polypedates macrotis* were accidentally found near the streamsides of Kalang river in the logged stands. Presumably they had moved from their original primary habitats and used streams as corridors, because their main habitats had been disturbed. It is possible that in these habitats, ecological conditions (like temperature and humidity) are relatively stable. The observation that arboreal or semi arboreal species (Table 8.26 and 8.27, Chapter VIII), and those other anurans living in streams, still survive in disturbed environments like in the logged stands supports the idea that they may adapt to altered environments.

Raymond and Handy (1991) found evidence of emigration and decreased survivorship in response to a logged site in a forest stream breeding pool. There was a displacement of individuals moving from the logged side of the breeding pool to the uncut side, and while survivorship remained relatively stable on the unaltered side, it decreased dramatically over a 3-year period for anurans that migrated to the pool from the logged site. A more important concern is that buffer strips could serve as corridors for amphibian movements. The consequences of corridors may be more serious for groups such as amphibians (in particular for non-riparian species) that undertake annual migrations between aquatic breeding sites and terrestrial home ranges.

The true riparian anurans of primary forest species such as *Rana laticeps, R signata* and *Staurois natator*, were found principally in the unlogged stands of the study sites, which was in line with results of an earlier study at a locality about 200km farther north in North Kalimantan (Inger 1987). These primary forest species were absent from the logged areas because their main habitats were primarily along clear, swift, rocky streams (Inger and Stuebing 1994). In contrast, *Rana limnocharis* and *R*

erythraea in the North Kalimantan forest were more prevalent in the logged stands than in primary forests, a similar pattern to that found in the present study.

Some of the observed patterns may not reflect responses to present conditions. Rather, they may represent the long-term consequences of logging whose immediate effects were not included in this study. For example, it was not possible to determine whether current populations of anurans in the logged sites have been maintained since logging, or represent recolonization. Certainly, past logging practices have produced habitat destruction of sufficient magnitude to extirpate resident populations of anurans (Burns 1972, Swanson and Lienkaemper 1978). Fragmentation and decreases in population size may decrease genetic variation, but Steven and Bruce (1988) found no evidence of this comparing populations of black bellied salamanders (*Desmognathus quadramaculatus*) in unlogged and logged watersheds in North Carolina.

If anuran populations are extirpated, the probability of recolonization varies among species. *Rana limnocharis* and *R erythraea*, the two disturbed-forest species are capable of long-distance terrestrial movements, but the true riparian species such as *Rana blythi*, *R laticeps* and *R signata* are not (Boeadi 1987). Anurans had not recovered on sites sampled several years after logging. It is not known how long a logged area requires to revert naturally to a condition in which the fauna typical of Kalimantan primary forest is regained. Further, it is also not known what can be done to expedite the recovery of these amphibian communities after logging.

Amphibians are often neglected in studies of ecosystems, but they are important both in terms of biomass and as possible useful indicator species for habitat disturbance. For example, Burton and Likens (1975) demonstrated that salamanders in a forest ecosystem may comprise a biomass twice that of birds and equal to mammals. Amphibians may be equally important to Kalimantan rain forest. In the study sites, roughly around 80 anurans were found in a 100m2 stream transect, but this figure probably underestimated the resident populations, since the sampling was not extensive enough to cover the whole of the streams in the study sites

9.3. Conclusions

The number of anurans obtained in pitfalls and searched quadrats was very low in that there was less than 1 individual anuran per pitfall trap or per quadrat searched either in the wet or dry season. The total number of species obtained either by pitfalls or by searched quadrat varied between 8-10 species during the wet season and around 7 species during the dry season at any site.

The results from searched quadrats and pitfall trapping indicated that the abundance of frogs was much lower in the open-canopy logged plots than in the closed-canopy logged and unlogged plots. The microclimatic components, in particular humidity and temperature, had significant effects on anuran abundance in the study sites. In addition, forest litter cover was significantly related to the abundance of anurans in all sites. This is likely to affect anurans especially the forest-dependent species , which so much depend on humidity, temperature and forest litter cover for their survival.

The proportions of forest-dependent anurans grouped by families recorded from pitfall traps and searched quadrats were represented most by pelobatids and microhylids in the unlogged and closed canopy logged plots. Whilst in the open canopy logged plots, the majority was represented by bufonids and ranids.

Leptobrachium abotti was the dominant pelobatid species obtained in the pitfalls and searched quadrats at any season in the unlogged plots, whilst *Rana limnocharis* was the dominant species observed in the open-canopy logged plots. *Leptobrachium*, as a forest litter-dependent anuran was completely absent in the open-canopy logged plots, and, in contrast, *R limnocharis* as a disturbed habitat species, was found very rarely in the primary forest of Unlogged Plots. No pelobatids were obtained either by pitfalls or by searched quadrat in open-canopy logged plots, indicating that this group of species may be considered as restricted to primary forest, unable to survive in disturbed habitat like the-open canopy logged plots, and would make an appropriate indicator species for disturbance.

The coefficient of similarity for species *Leptobrachium abotti, Kalophrynus plurostigma* and *Bufo biporcatus* in the pitfall and searched quadrats was very high in the wet and dry seasons because all these species are forest litter species and occur together at ground level in the observed plots of unlogged and closed canopy logged plots. In contrast, the coefficient of similarity between the forest litter species and *Rana chalconota* and *R. limnocharis* was very low, because they did not occur in the same level of habitat.

A total of 43 anuran species was recorded from night riparian transects surveyed in Kalang forest streams at Camp 92, Central Kalimantan. These species may be grouped into four general ecological types. The mean numbers of individual anurans and anuran species richness varied slightly between sites at any season. The variation may reflect seasonal behaviour of each species that is only temporary.

The presence and absence of a riparian species is as likely to be determined by the physical characteristics of a stream as by the presence of ecologically related species. That is, the physical characteristics of streams determine the success of particular ecological types, each of which may include a number of species.

Abundance of anurans was significantly different between streams flowing through the unlogged and logged plots. The species composition and richness of the anurans were also significantly different between streams at all sites. One major physical characteristic that differs between streams in the Unlogged and Logged Plots was that Logged Plots had generally finer (silt) sediments. Logging and associated road construction resulted in increased sediment loads in the logged forest areas and most of the increase of fine sediment observed in the logging sites was derived from logging practices. The increased sedimentation apparently caused habitat destruction through filling of substrate crevices and reduced the anuran abundance.

Canopy cover, air humidity and temperature in the Unlogged Plots were also different from the Logged Plots. The difference was mainly because intensive selective logging opens primary forest canopy and creates large canopy gaps. As a result, the open areas had greater daily fluctuation in temperature and humidity compared to the primary

forest. The mixture of these two important factors affects anuran abundance. The forest canopy cover along the streams had an important function, not only to maintain humidity but also to reduce water loss, which is important for anurans, particularly during dry periods when water in the forest is very limited.

Concluding remarks

Recently, 600-ha Permanent Sample Plots have been established in the Kayu Mas Forestry Logging Concession (180.000ha) as a model to investigate the impact of logging on Kalimantan biodiversity. This forestry concession had been logged by the Indonesian selective logging system with little or no regard for the unique wildlife communities of typical Kalimantan forest associated with Dipterocarp primary forest. Large block cuts left no shade or islands of natural forest (to provide reservoirs for species reinvasion) and the terrain is scarred and strewn with logging debris.

The present study has concluded that logging had a significant effect on the abundance of anurans, species similarity and composition in the study sites. Although anurans are the most abundant vertebrate of the forest ecosystems in particular for the Kalimantan forest, their populations cannot be managed properly because knowledge of the unique anurans of the Kalimantan forest is still very limited. In addition, how logged wasteland can be restored to suitable habitat for forest dependent anurans is still in question.

9.4. Challenge for Future Research

A challenge to investigations of amphibian-forestry relationships is the decision on which measurement variables are most appropriate for characterizing a site's population status. For example, is density (or an index thereof, such as relative abundance) a good indicator of habitat quality for amphibians? A further complication is the necessity of deciding how long a sampling period is needed to be confident about (i) local richness estimates (the detection of rare species requires disproportionately greater sampling effort than common species; Buchlmann *et al.* 1993) and (ii) site-specific population size inferences. Questions about the appropriate management variable and sampling frame depend in large part on the population

stability of the specific amphibian species being sampled, because an inherently fluctuating population can lead to unreliable conclusions based on density estimates (Van Horne 1983) and (or) misleading inferences about population trends and their causes (Peachmann *et al.* 1991).

The understanding of the impacts of forest management practices on wildlife is still evolving and it is only quite recently that nongame taxa other than birds have received recognition by forest management agencies (Szaro 1988). An analysis by Gibbons (1988) of the research focus by major publications in the fields of general and applied ecology during the 1980s found that amphibians were the least represented (<5%) among vertebrate classes in the literature. Most of the current knowledge regarding amphibian relationships to forestry is based on studies published during the last 10 years, and the demand for more empirical information is growing as foresters increasingly incorporate wildlife conservation objectives into their long-term management plans. Therefore, the discussion of a few major unstudied aspects of amphibians and forestry practices may help stimulate future research in areas where more information is critically needed.

9.5. Recommendation

Accumulating evidence suggests that some forestry practices can have a negative impact on habitat quality for certain species, but maintaining sustainable anuran populations can generally be compatible with timber harvesting if specific precautions are taken. When considering amphibian conservation during forestry management practices, a fundamental challenge will be to minimize differences in quantity and quality of forest floor microhabitats between recently logged stands and stands regenerated from a recent natural disturbance. The implicit assumption is that amphibians and other organisms have evolved adaptations to natural disturbances and thus will be relatively tolerant of logging and other timber management practices if they resemble a natural disturbance. Despite this optimism about the compatibility of forest logging and amphibian habitat, a conservative approach to maintaining viable populations of amphibians would also ensure that a representative array of forest types remained in unharvested state. For example, a total buffer strip width ranging from 30 (Rudolph and Dickson 1990) to 100m (McCombe *et al.* 1993) along stream courses

was recommended for mitigating the effects of logging on stream amphibians. A buffer strip width of 30 metres, provides roughly 60% canopy cover (Steinblums *et al.* 1984), which could help maintain moist, cool conditions for terrestrial amphibians. Buffer strips along stream courses benefit stream amphibians by preventing the accumulation of high silt concentrations and by maintaining cool temperatures (Corn and Bury 1989, Bury *et al.* 1991, Welsh and Lind 1991). Streamsides should also be protected as vital to amphibians because they harbor high densities of both aquatic- and terrestrial-breeding species (Corn and Bury 1989). Moreover, stream banks may be potential refuge sites for terrestrial-breeding amphibians in the face of large-scale disturbances (Bury 1988), they may be a source for the repopulation of harvested stands (Welsh 1990), and they may facilitate species' dispersal into these marginal habitats. Because logging has a long-term negative effect on all amphibian species (Corn and Bury 1989), a buffer zone should be left untouched on either side of all streams (Franklin *et al.* 1981).

Tracts of primary forest adjacent to logged forest should be protected, because the primary forests can serve as a species pool for the recolonization of marginal, post-harvest stands. Similarly, forested corridors along streamsides could significantly enhance the effectiveness of primary forest in sustaining habitat for amphibians.

The next generation of forestry-amphibian investigations should focus as much on evaluating silvicultural prescriptions that successfully mitigate the effects of logging on amphibians as the previous generation has on documenting the negative magnitude of effects. Those researching the relationship between amphibians and forestry practices should apply their results, and give informative explanations to forest managers and concessionaires about how to minimize logging disturbance, so that the integrity of the forest ecosystem is maintained along with healthy animal populations, including amphibians. In addition, future studies that integrate site-specific ecological data and information on anuran population structure are needed to evaluate the potential long-term viability of anuran species and to make more effective forest-level management recommendations.

REFERENCES

Abdulhadi R. (1975) A review on Southeast Asia logging industries. *Asian Forestry Review*, **31**, 47-51.

Abdulhadi R. (1994) The planning and design of minimizing the impact of logging in Bornean tropical forest. Unpubl. report to Indonesian Ministry of Forestry, Jakarta. Indonesia.

Abdulhadi, R. Kuswata K., and Sukardjo S. (1981) Effects of mechanized logging in the lowland dipterocarp forest at Lempake, East Kalimantan. *Malay. For.*, **44** (2-3), 407-418.

Aber, J.D., Botkin, D.B., and Melillo, J.M. (1978) Predicting the effects of different harvesting regimes on forest floor dynamics in northernwoods. *Can. J. For. Res.*, **8,** 306-315.

Achmad, A. (1988) Observations on feeding ecology of Crested Toad, *Bufo divergens* in Khayan streams, Central Kalimantan. Field Report LIPI-WWF, LWP IV/ 4. Jakarta. Indonesia.

Achmad, M. (1982) Assessment of anurans abundance in the unlogged and logged stands (by pitfall trapping) in Dipterocarp forests of North Kalimantan. Paper presented at the Seminar Biodiversity in Kalimantan production forests, 15-18 July 1995, Jakarta. Indonesia.

Adam, J.N., and Beschta, R.L. (1980) Gravel bed composition in Oregon coastal streams. *Can. J. Fish. Aquat. Sci.*, **37**, 1514-1521.

Alexander, T.R. (1964) Observations on the feeding behavior of *Bufo marinus* (Linne). *Herpetologica*, **20**, 255-259.

Alvarado, R. H. (1967) The significance of grouping on water conservation in Ambystoma, *Copeia*, **1967**, 667-668.

Andreone, F. (1994). The amphibians of Ranomafana rain forest, Madagascar - preliminary community analysis and conservation considerations. *Oryx*, Vol. **28** No. 3. July 1994.

Andrus, C. and Froehlich, H.A. (1988) Streamside forest development after logging or fire in the Oregon Coast Range: wildlife habitat and timber value. *In* K.J. Raedeke (Ed.). Streamside Management: Riparian Wildlife and Forestry Interactions. Proc. Symp. 11-13 February 1987, Univ. of Washington, Seattle. Institute of Forest resources, Univ. of Washington, Contrib. 59, pp. 139-152.

Anonymous (1993) Physical characteristics of tropical rain forests in Kalimantan. Report V, Bappeda Jakarta. Indonesia.

Anonymous (1994) The EIA requirements for a sustainable development of Indonesian rain forests. Directorate General of Forest Utilization. Jakarta. Indonesia.

Anonymous (1995) Effects of selective felling on dipterocarp lowland rainforest in Central Borneo. Unpublished report to UK-Indonesia Tropical Forest Management Project, Jakarta. Indonesia.

Ash, A.N. (1988) Disappearance of salamanders from clearcut plots. *J. Elisha Mitchell Sci. Soc.*, **104**, 116-122.

Ash, A.N., and Bruce, R.C. (1994). Impacts of timber harvesting on salamanders. *Conserv. Biol.*, **8**, 300-301.

Ashton, P.M.S. (1992) Some measurements of the microclimate within a Sri Lankan tropical rain forest. *Agricultural and Forest Meteorology*, **59**, 217-235.

Aubry, K.B., and Hall, P.A. (1991) Terrestrial amphibian communities in the southern Washington Cascade Range. *In* Ruggiero, L.F., Aubry, K.B., Carey, A.B., and M.H. Huff. Wildlife and vegetation of unmanaged Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-285. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, pp. 326-338.

Aubry, K.B., Jones, L.L.C., and Hall, P.A. (1988) Use of woody debris by plethodontid salamanders in Douglas-fir forests in Washington. *In* Management of amphibians, reptiles, and mammals in North America. (R.C. Szaro, K.E. Severson, and D.R., Patton, Eds.). USDA For. Serv. Gen. Tech. Rep. RM-166, pp. 32-37.

Baker, A (1984) Timber, the environmental and wildlife in Neotropical and African rain forests. Paper presented in the World Forestry Congress, Bali, Indonesia.

Barbault, R.A. (1974) The feeding diets of African anurans in the savanna and forest habitats in the Ivory Coast, Africa (English version). Bull. Inst. *France Afrique Noire*, **36**: 952-772.

Barbault, R.A. (1984) Community structure and reproductive ecology of *Rana calamitants* in South America. *American J.*, **2**, 45-58.

Barinaga, M.(1990) Where have all the froggies gone? *Science*, **247**, 1033-1034.

Beauregard, N., and Leclair, R. Jr. (1988) Multivariate analysis of the summer habitat structure of *Rana pipiens* Schreber. *In* Lac Saint Piere (Quebec, Canada). R.C. Szaro, K.E. Severson, and D.R. Patton (eds.). Management of Amphibians, Reptiles, and Small Mammals in North America. U.S. Department of Agriculture, Forest Service, General Technical Report RM-166, pp. 129-143.

Becker, P., Erhart D.W., and Smith A.P. (1989) Analysis of forest light environments. Part I. Computerised estimation of solar radiation from hemispherical canopy photographs. *Agricultural and Forest Meteorology*, **59**, 217-235.

Bennett, S.H., Gibbons J.W., and Glanville J. (1980) Terrestrial activity, abundance and diversity of amphibians in differently managed forest types. *Amer. Midl. Nat.*, **103**, 412-416.

Berry, P.Y. (1966) The food and feeding habits of the torrent frog, *Amolops larutensis*. *J. Zool. London*, **149**, 204-214.

Berry, P.Y. (1975) The Amphibian Fauna of Peninsular Malaysia. Tropical Press, Sabah, Malaysia.

Beschta, R.L. (1978) Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. *Water Resour. Res.*, **14**, 1011-1016.

Beschta, R.L., Bilby, R.E. Brown, G.W., Holtby L.B., and Hofstra, T.D. (1987). Stream temperature and aquatic habitat: fisheries and forestry interactions. *In* E.O. Salo and T.W. Cundy (Eds.), Streamside Management: Forestry and Fishery Interactions. Proc. Symp., 12-14 February 1986, Univ. of Washington, Seattle. Institute of Forest Resources, Univ. Washington, Contrib. 57, pp. 199-232.

Bharata, T. (1989) Observations of some anuran species in the secondary forest of Central Kalimantan. Unpublished report. University of National, Jakarta. Indonesia.

Bishop, C.A., and Petit K.E. (1992) Declines in Canadian amphibian populations: designing a national monitoring strategy. Occasional paper no. 76. Canadian Wildlife Service, Ottawa.

Blaustein, A.R. (1994) Amphibian declines: judging stability, persistence and susceptibility of populations to local and global extinctions. *Conservation Biology*, **8**, 60-71.

Blaustein, A.R. and Wake D.B. (1990) Declining amphibian populations: a global phenomenon? Trends in *Ecology and Evolution*, **5**(7), 203-204.

Blaustein, A.R., and Olson D.H. (1991). Declining amphibians. *Science*, **253**, 1467.

Blaustein, A.R., Hokit, D.G., and O'Hara R.K. (1994) Pathogenic fungus contributes to amphibian losses in the Pacific Northwest. *Biological Conservation*, **67**, 251-254.

Blymyer, M.J., and McGinnes, B.S. (1977) Observations on possible detrimental effects of clearcutting on terrestrial amphibians. *Bull. Md. Herpetol. Soc.*, **13**, 79-83.

Boeadi, H. (1990) Ecological and competitive relation among some riparian species in Barito River, South Kalimantan. *Zoology Bulletin*, **5**, 51-59.

Bonin, J. (1991) Effect of forest age on woodland amphibians and the habitat and status of stream salamanders in southwestern Quebec. M.S. Thesis, Department of Renewable Resources. McGill University, Montreal, P.Q.

Borhan, B.M., Johari B.B. and Quah E.S. (1989) Studies on logging damage due to different methods and intensities of harvesting in Tekam Forest Reserve. Interim report to Kementerian Kehutanan Pahang, Peninsular Malaysia.

Bormann, F.H. and Likens, G.E., Siccama, T.G. Pierce, R.S., and Eaton J.S. (1974) The export of nutrients and recovery of stable conditions following deforestation at Hubbard Brook. *Ecol. Monogr.*, **44**, 255-277.

Bormann, F.H., and Likens, G.E. (1979) Pattern and process in forested ecosystems. Springer-Verlag, New York.

Boulenger, G.A. (1912) A vertebrate fauna of the Malay Peninsula from the Isthmus of Kra to Singapore including the adjacent islands. Reptilia and Batrachia. Taylor and Francis, London.

Bradford, D.F., Graber, D.M., and Tabatabai F. (1994) Population declines of the native frog, *Rana muscosa*. *In* Sequoia and Kings Canyon National Parks, California. *Southwestern Naturalist*, **39**, 323-327.

Branch, W.R. (1976) Two exceptional food records for the African bull frog *Pyxicephalus adspersus* (Amphibia, Anura, Ranidae). *J. Herpetol.*, **10**, 266-268.

Brattstorm, B.H. (1963) A preliminary review of the thermal requirements of amphibians. *Ecology*, **44**, 238-255.

Brattstorm, B.H.. (1968) Thermal acclimation in anuran amphibians as a function of latitude and altitude. *Comp. Biochem. Physiol.*, **24**, 93-11.

Bray, J.R., and Gorham E. (1964) Litter production in the forests of the world. *In* J.B. Cragg (Ed.). *Advances in Ecological Research*, **2**, 101-157.

Brodie, E.D., Jr., and Howard, R.R. (1973) Experimental study of Batesian mimicry in the salamanders *Plethodon jordani* and *Desmognathus ochrophaeus*. *Am. Midl. Nat.*, **90**, 38-46.

Brodie, E.D., Jr., Nowak, R.T., and Harvey W.R. (1979) The effects of antipredator secretions and behavior of selected salamanders against shrews. *Copeia*, **1979**, 270-274.

Brokaw, N.V.L. (1985) Gap phase regeneration in a tropical forest. *Ecology*, **66**, 682-687.

Brown, G.B, Kim Y.H., Mosher H.S., Fuhrman G.J., and F.S. Fuhrman (1977) Chemistry and physiology of skin from the frog *Atelopus zeteki*. *Biol. Bull.*, **109**, 238-254.

Brown, N. (1993) The implications of climate and gap microclimate for seedling growth conditions in a Bornean lowland rain forest. *J. Trop. Ecol.*, **9**, 153-168.

Bryant, , M.D. 1985. Changes 30 years after logging in large woody debris, and its use by aquatic communities. *In* R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Fflolliot and R.H. Hamre (Technical Coordinators), Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. Proc. First North American Riparian Conference, 16-18 April 1985, at Tuscon, AZ. USDA For. Serv. Gen. Tech. Rep. RM-120, pp. 329-334.

Buenoflor, M., and M.H. Tiki (1989) Impact of selective logging on habitat destruction in New Guinean tropical forest. Unpublished report, Forestry Service Agency of Papua New Guinea.

Buhlmann, K.A., Mitchell, J.C., and Pague, C.A. (1993) Amphibian and small mammal abundance and diversity in saturated forested watershed wetlands and adjacent uplands of southeastern Virginia. *In* Proceedings of a Workshop on Saturated Forested Wetlands in the Mid-Atlantic Region: The State of the Science. (S.D. Eckles, A. Jennings, A. Spingarn, and C. Wienhold., Eds.). U.S. Fish and Wildlife Service. Annapolis, Md., pp 1-7.

Burgess, P.F. (1971) Effect of Logging on hill dipterocarp forest. *Malayan Nature Journal*, **24**, 231-237.

Burns, J.W. (1972) Some effects of logging and associated road construction on northern Carolina streams. *Rans. Am. Fish. Soc.*, **101**, 1-17.

Burton, T.A and Likens E.G. (1975) Salamander populations and biomass in the Hubbard Brook Experimental Forest, New Hampsire. *Copeia*, **1975**, 541-54.

Burton, T.A. (1976) An analysis of the feeding ecology of the salamanders (Amphibia, Urodela) of the Hubbard Brook Experimental Forest, New Hampshire. *J. Herpetol.*, **10**, 187-204.

Bury, R. B. (1983) Differences in amphibian populations in logged and old growth redwood forest. Northwest. *Science*, **57**, 167-178.

Bury, R. B. (1988) Habitat relationships and ecological importance of amphibians and reptiles. *In* Raedke, Kenneth J. (Ed.). Streamside management: riparian wildlife and forestry interactions. Inst. Resour. Contrib. 59. Seattle, WA: University of Washington, pp. 61-76.

Bury, R. Bruce (1972) Small mammals and other prey in the diet of the Pacific giant salamander (*Dicamptodon ensatus*). *Amer. Midl. Nat.*, **87**, 524-526.

Bury, R.B. and Corn, P.S. (1987) Evaluation of pitfall trapping in northwestern forests: trap arrays with drift fences. *J. Wild. Manag.*, **51**, 112-119.

Bury, R.B. and Corn, P.S. (1988) Responses of aquatic and streamside amphibians to timber harvest: a review. *In*: Raedke, Kenneth J. (Ed.). Streamside management: riparian wildlife and forestry interactions. Inst. Resour. Contrib. 59. Seattle, WA: University of Washington, pp. 61-76.

Bury, R.B. and Corn, P.S. (1991) Wildlife-Habitat Relationships: Sampling Procedures for Pacific Northwest Vertebrates. *In* A.B. Carey and L.F. Ruggiero (Eds.). USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon. General Technical Report PNW-GTR-275, pp. 1-27.

Bury, R.B., Campbell, H.W., and Scott, N.J. Jr. (1980) Role and importance role of nongame wildlife. Trans. N. Amer. Wildlife and Natr. Res. Conf. 45, pp. 197-207.

Campbell, H.W., and Christman, S.P. (1982) Field techniques for herpetofaunal community analysis. *In* N.J. Scott, Jr. (Ed.), Herpetological Communities. USDI, Fish and Wildl. Serv., *Wildl. Res. Rep.*, **13**, 193-200.

Campbell, I.C., and Doeg, T.J. 1989. Impact of timber harvesting and production on streams: a review. *Aust. J. Mar. Freshwater Res.*, **40**, 519-539.

Canham, C.D., Denslow, J.S., Platt, W.J., Runkle, J.R., Spies, T.A. and White, P.S. (1990) Light regimes beneath closed canopies and tree-fall gaps in temperate and tropical forests. *Canadian Journal of Forest Research*, **20**, 620-631.

Cannon, C.H., Peart D.R., Leighton, M., Kartawinata, K. (1994) The sturcture of lowland rain forest after selective logging in West kalimantan, Indonesia. *Forest Ecology and Management*, **67**, 49-68.

Carey, C. (1978) Factors affecting body temperatures of toads. *Oecologia*, **35**, 197-219.

Carey, C. (1993) Hypothesis concerning the causes of the disappearance of boreal toads from the mountains of Colorado. *Conservation Biology*, **7**, 355-361.

Carey, C., and Bryant C.J. (1995) Possible interrelations among environmental toxicants, amphibian development, and decline of amphibian populations. *Environmental Health Perspective* **103**(4), 13-17.

Cei, J.M. (1969) The patagonian telmatobiid fauna of the volcanic Somuncura Plateau. J. *Herpetol.*, **3**, 1-18.

Chandrashekara, U.M., and Ramakrishnan P.S. (1993) Gap phase regeneration of tree species of differing successional status in a humid tropical forest of kerala, India. *J. Biosciences*, **18**, 279-290.

Chew, W.L. (1968) Conservation of habitats. *In* Conservation in Tropical South East Asia. *IUCN Publ. New Ser.*, **10**, 337-339.

Christian, K.A. (1982) Changes in food niche during post-metamorphic ontogeny of the frog *Pseudacaris triseriata*. *Copeia*, **1982**, 73-80.

Christman, S.P., Kochman, H.I., Campbell, H.W., Smith, C.R., and W.S. Lippincott. Jr. (1979) Successional changes in community structures: amphibians and reptiles in Florida. USDI, Fish and Wildl. Serv., Natl. Fish and Wildl. Lab., Gaines ville, FL, 21pp.

Clarkson, R.W., and Rorabaugh, J.C. (1989) Status of leopard frogs (*Rana pipiens* complex: Ranidae) in Arizona and southeastern California. *Southwestern Naturalist*, **34**, 531-538.

Claussen, D.L. (1969) Studies of water-loss and rehydration in anurans. *Physiol. Zool.*, **42**, 1-14.

Clawson, R.G., Lockaby, B.G., and Jones, R.H. (1996) Amphibian responses to harvesting in forested floodplains of low order, blackwater streams. *For. Ecol. Manage.*, In press.

Collins, J.P. (1975) *A Comparative Study of the Life History Strategies in a Community of Frogs*. Ann Arbor: Univ. Michigan, PhD. Dissertation.

Collins, N.M., Sayer, J.A. and Whitmore, T.C. (1991) *The Conservation Atlas of Tropical Forests*: Asia and the Pacific. Macmillan, London.

Connell, J.H. (1961) The influence of interspecific competition and other factors on the distribution of the barnacle Chthamalus stellatus. *Ecology*, **42**, 710-723.

Conner, E.J., Trush, W.J. and Knight, A.W. (1988) Effects of logging on Pacific giant salamanders: influence of age-class composition and habitat complexity. *Bull. Ecol. Soc. Am.*, **69** (suppl.), 104-105.

Cordero, A. (1987) Inventory of tropical amphibian assemblages in African forest. *Ecology*, **37**(3), 484-495.

Corn, P.S., and Bury R.B. (1991) Terrestrial amphibian communities in the Oregon Coast Range. *In* Ruggiero, L.F.; Aubry K.B.; Carey A.B., and M.H. Huff. Wildlife and vegetation of unmanaged Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-285. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, pp. 304-317.

Corn, P.S., and Bury, R.B. (1989) Logging in western Oregon: responses of headwater habitats and stream amphibians. *Forest Ecology and Management*, **29**, 39-57.

Corn, P.S., and Bury, R.B. (1990) Sampling methods for terrestrial amphibians and reptiles. Gen. Tech. Rep. PNW-256. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, 34 p.

Corn, P.S., and Bury, R.B. (1993) Large declines of toads (Bufo) at different geographic scales: coincide or coordination? Unpublished abstract, submitted to Second World Congress of Herpetology. Adelaide, Australia.

Costa, R., Flolliott P.F., and Patton, D.R. (1976) Cottontail responses to forest management in southwestern panderosa pine. U.S.D.A. Forest Serv. Research Note RM-330.

Crome, F.H.J. (1991) Wildlife Conservation and rain forest management - examples from northeast Queensland, in Rain Forest Regeneration and Management (A. Gomez-Pompa, T.C. Whitmore and M. Hadley, Eds.), Parthenon Press, Carnforth and Paris, pp. 407-416.

Crump, M.L. (1971) Quantitative analysis of the ecological distribution of a tropical herpetofauna. *Occ. Pap. Mus. Nat. Hist. Univ. Kansas*, **3**, 1-62.

Crump, M.L. (1986) Qualitative analysis of the ecological distribution of a tropical herpetofauna. *Occas, Pap. Mus. Nat. Hist.* Univ. Kansas, **3**, 1-61.

Dalrymple, G.H. (1988) The herpetofauna of Long pine Key, Everglades National Park, in relation to vegetation and hydrology. *In* R.C. Szaro, K.E., and D.R. Patton (Eds.). Management of Amphibians, Reptiles, and Small Mammals in North America, U.S. Department of Agriculture, Forest Service, General Technical Report RM-166, pp. 72-86.

Dawkins, H.C. (1959) The volume increment of natural tropical high forest and limitations on its improvement . *Empire Forestry Review*, **38**, 175-80.

De Graaf, N.R. (1986) *A Silvicultural System for Natutal Regeneration of Tropical Rain Forest in Suriname*. Agricultural University, Wageningen.

De Graaf, N.R., and Rudis, D.D. (1990) Herpetofaunal species composition and relative abundance among three New England forest types. *For. Ecol. Manage.*, **32**, 155-165.

De Graaf, R.M., and Yamasaki, M. (1992) A nondestructive technique to monitor the relative abundance of terrestrial salamanders. *Wildl. Soc. Bull.*, **20**, 260-264.

Diller, L.V., and Wallace, R.L. (1994) Distribution and habitat of *Plethodon elongatus* on managed young growth forests in north coastal California. *J. Herpetol.*, **28**(3), 310-318.

Dimmitt, M.A., and Ruibal R. (1980) Exploitation of food resources by spadefoot toads (*Scaphiopus*). *Copeia*, **1980**, 854-862.

Directorate General of Forest Utilization (1987). Management and Conservation for Sustainable Forest of Indonesia. Proceedings of Biodiversity Conservation Management Workshop in Puncak, Bogor,Indonesia 5-8 July 1987, pp 93-98.

Djoko, S. (1989) Ecological study on community structure of *Rana macrodon*, *R blythi and Rana ibanorum* in a small rainforest stream of North Kalimantan. *Journal of Herpetofauna*, **3**, 41-45.

Dodd, K.C., Jr. (1991) The status of red Hills salamander Phaeognathus hubrichti. Alabama. USA. 1976-1988. *Biol. Conserv.*, **55**, 57-75.

Douglas, I. (1968) Erosion in the S. Gombak catchment, Selangor, Malaysia. *J. trop. Geogr.*, **26**, 1-16.

Douglas, I., Spencer, T., Greer T., and Meng W.W. (1992) The impact of selective commercial logging on stream hydrology, chemistry and sediment loads in the Ulu Segama rain forest, Sabah, Malaysia. *Phil. Trans. R. Soc. Land. B.*, **335**, 397-406.

Drost, C.A., and Fellers G.M. (1993) Decline of frog species in the Yosemite section of the Sierra Nevada. Report to Yosemite National Park and the Yosemite Association. 59 pp.

Duellman, W.E., and Linda Trueb (1994). Biology of Amphibians. The Johns Hopkins University Press, London.

Duelmann, W.E. (1960) A distributional study of the amphibians and reptiles of the Isthmus of Tehuantepec, Mexico. *Univ. Kansas Publ. Mus. Nat. Hist.*, **13**, 19-72.

Duelmann, W.E. (1978) The biology of an equatorial herpetofauna in Amazonian Ecuador. *Misc. Publ. Mus. Nat. Hist. Univ. Kansas*, **65**, 1-352

Duncan, S.H., and Ward J.W. (1985) The influence of watershed geology and forest roads on the composition of salmon spawning gravel. *Northwest Sci.*, **59**, 204-212.

Dunson, W.A., Wyman, R.L., and Corbett, E.S. (1992) A symposium on amphibian declines and habitat acidification. *J. Herpetol.*, **26**(4): 349-352.

Dupuis, L.A., Smith, J.N.M., Bunnell, F. (1995) Relation of terrestrial-breeding amphibian abundance to tree-stand age. *Conserv. Biol.*, **9**(3), 645-653.

Edenham, P. (1996) *Spatial dynamics of the European treefrog (Hyla arborea L.) in a heterogenous landscape*. Doctoral dissertation, Swedish University of Agricultural Sciences, Uppsala, Sweden.

Ehrlich, P.R. and Ehrlich A.H. (1981) *Extinctions*. Random House, New York.

Elliot, A.B., and Karunakaran L. (1974) Diet of *Rana cancricora* in fresh water and brackish water environments. *J. Zool.* (London), **174**, 203-215.

Emlen, J. (1973) Ecology: an evolutionary approach. Addison-Wesley Publishing Company, Reading, Mass. 493 pp.

Enge, K.M. and Marion W.R. (1986) Effects of clearcutting and site preparation on herpetofauna of a North Florida Flatwoods. *Forest Ecology and Management*, **14**, 177-192.

Ernst, C.H., and Barbour, R.W. (1989) Snakes of Eastern North America. George Mason University Press, Fairfax. Va. 282 pp.

Ewel, J. and Conde L. (1976) Potential ecological impact of increased intensity of tropical forest utilization. Unpublished report to USDA Forest Service, Madison, Wisconsin, USA.

FAO (1988) *An Interim Report on the State of Forest Resources in the Developing Countries*. Forest Resource Division, Forest Dept. FO:MISC/88/7. FAO, Rome.

Fauth, H., Herrington R.E. and Henderson B.A. (1989) Reproductive ecology and systematics of salamanders of the genus Batrachoseps. *Zoology*, **54**, 1-46.

Feder, M.E. (1982) Thermal ecology of Neotropical lungless salamanders (Amphibia: Plethodontididae): environmental temperatures and bevavioral responses. *Ecology*, **63**, 1665-1674.

Feder, M.E. (1983) Integrating the ecology and physiology of plethodontid salamanders. *Herpetologica*, **39**, 291-310.

Feder, M.E., and Lynch, J.F. (1982) Effects of latitude, elevation, and microhabitat on field body temperatures of neotropical and temperate zone salamanders. *Ecology*, **63**, 1657-1664.

Fellers, G.M., and Drost C.A. (1993) Decline of Anurans in Western U.S. National Parks. Unpublished abstract, submitted to Second World Conggress of Herpetology in Australia.

Fitch,H.S. (1965) The University of Kansas Natural History Reservation in 1965. *Misc. Publ. Mus. Nat. Hist. Univ. Kansas*, **42**, 1-60.

Foley, D.H. (1994) *Short-term response of herpetofauna to timber harvesting in conjunction with streamside-management zones in seasonally flooded bottomland-hardwood forests of southeast Texas*. M.S. Thesis. Texas A&M University. Texas.

Forest Management Bureau (1988*) Natural Forest Resources of the Philippines*. Philippine-German Forest Resource Inventory Project. (pp. 62 mimeo).

Franz, R., and Ashton, R.E. Jr. (1989) Behavior and Movements of Certain Small Sandhill Amphibians and Reptiles in Response to Drift Fences. Unpublished Report. Florida Game Fresh Water Fish Commission, Nongame Progra,, Tallahassee, Florida.

Fraser, D.F. (1976) Emperical evaluation of the hypothesis of food competition in salamanders of the genus Plethodon. *Ecology*, **57**, 459-471.

Freed, A.N. (1980). Prey selection and feeding behavior of the green treefrog (Hyla cinerea). *Ecology*, **61**, 461-465.

Freezailah Che Yeom (1984) Lesser-known tropical wood species: how bright is their future? *Unasylva*, **36**(145), 3-16.

Friend, G.R. (1984) Relative efficiency pf two pitfall-drift fence systems for sampling small vertebrates. *Australian Zoologist*, **22**, 423-433.

Frisbie, M.P. and Wyman, R.L. (1991) The effect of soil chemistry on sodium balance in the red-backed salamander, *Plethodon cinereus*, and three other salamanders. *Physiol. Zool*, **64**(4), 1050-1068.

Frisbie, M.P. and Wyman, R.L. (1992) The effects of soil pH on sodium balance in the red-backed salamander, *Plethodon cinereus*: A comparison of two forest types. *J. Herpetol*, **26**(4), 434-442.

Fromm, P.O. (1956) Heat production of frogs. *Physiol. Zool.*, **29**, 234-240.

Gascon, C. (1991) Population-and community-level analysis of species occurence of Central Amazonian Rainforest tadpoles. *Ecology*, **72**(5), 1731-1746.

Gause, G.F. (1934) The struggle for existence. Williams and and Wilkins, Baltimore. 163 p.

Gibbon, J.W. 1988. The management of amphibians, reptiles, and small mammals in North America: the need for an environmental attitude. *In* Management of amphibians, reptiles and mammals in North America. (R.C. Szaro, K.E. Severson, and D.R. Patton, Eds.). USDA For. Serv. Gen. Tech. Rep. RM-166. pp. 4-10.

Gibbons, J.W., and Semlitsch, R.D. (1982) Terrestrail drift fences with pitfall traps: An effective technique for quantitative of animal populations. *Brimleyana*, **7**, 1-16.

Gilbert, F.F. and Allwine, R., R. (1991) Terrestrial amphibian communities in the Oregon Cascade range. *In*: Ruggiero, L.F., Aubry, K.B., Carey, A.B., and M.H. Huff (Eds.). Wildlife and vegetation of unmanaged Douglas-fir forests. Gen. tech. Rep. PNW-GTR-285. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, pp. 318-324.

Grant, B.W., Brown, K.L. and Gibbons, J.W. (1994) Changes in amphibian biodiversity associated with 25 years of pine forest regeneration: implications for biodiversity management. *In* Biological diversity: problems and challenges,(S.K. Majumdar, F.J. Brenner, J.E. Lovich, J.F. Scahlles, and E.W. Miller, Eds). The Pennsylvania Academi of Science, Philadelphia. Pa., pp. 355-367.

Grizzell, R.A. Jr. (1949) The hibernation site of three snakes and a salamander. *Copeia*, **1949**, 231-232.

Hairston, N.G. (1949) The local distribution and ecology of the plethodontid salamanders of the southern Appalachians. *Ecological Monographs*, **19**, 47-73.

Hairston, N.G. (1981) An experimental test of guild: salamander competition. *Ecology*, **62**, 65-72.

Hairston, N.G. (1989) Ecological experiments. Cambridge University Press. Cambridge.

Hairston, N.G. 1980. Species packing in the salamander genus Desmonagthus: What are the interspecific interactions involved? *American naturalist* **115**, 354-366.

Hairston, N.G., Sr. (1987) Community ecology and salamander guilds. Cambridge Univ. Press, Cambridge, Great Britain, 230 pp.

Hall, R.J. 1980. Effects of environmental contaminants on amphibians and reptiles: a review. US. Dep. Inter., Fish and Wildll. Serv. Spec. Sci. Rep.228. Washington, D.C.

Hamilton, L.S. 1985. Some water/soil consequences of modifying tropical rain forests. *In: The Future of Tropical Rain Forests in South East. Ecology*, **10**, 69-80.

Hamzah Z. (1978). Some observations on the effects of mechanical logging on regeneration, soil and hydrological conditions in East Kalimantan. *Biotrop Special Publication*, **3**, 73-8.

Hamzah Z. (1985) The development of Lesser-known tropical forest species in Indonesia. *Kalawarta,* **2**, 25-30

Harmelien-Vivien, M.L. & F. Bourliere. 1989. *Vertebrates in complex tropical systems.* New York: Springer-Verlag.

Hassinger, J.S., Lisconsiky, S.A., and Shaw, S.P. (1975) Wildlife in Clearcutting in Pennysilvania. School of Forest Resources, The Penna. State Univ., pp. 65-81.

Hawkins, C.P. Murphy, M.L., Anderson N.H., and Wilzbach M.A. (1983) Density of fish and salamanders in relation to riparian canopy and physical habitat in streams of the northwestern United States. *Can. J. Fish. Aquat. Sci.*, **40**,1173-1185.

Hawthorne, R. (1991) Natural forest management by the Yanesha Forestry Cooperative in Ghana. *In* Alternatives to Deforestation (A.B. Anderson, Ed.), Columbia University Press, New York, USA.

Hayes, M.P., and M.R. Jennings (1986) Decline of ranid frog species in western North America: Are bullfrogs (Rana catesbeiana) responsible: *J. Herpet.* **20**(4), 490-509.

Heatwole, H. (1960) Burrowing ability and behavioral responses to desiccation of the salamander, *Plethodon cinereus*. *Ecology*, **41**, 661-668.

Heatwole, H. (1962) Environmental factors influencing local distribution and activity of the salamander. *Plethodon cinereus*. *Ecology*, **43**, 460-472.

Heatwole, H., Cameron, E., and Webb, G.J.W. (1971) Studies on anuran water balance. II. Evaporative water loss, vital limit, and behavioral responses to desiccation in Notaden bennetti. *Herpetologica*, **27**, 365-378.

Heatwole, H.E., and Sexton O.J. (1966) Herpetofaunal comparisons between two climate zones in Panama. *Am. Midl. Nat.*, **75**, 45-60.

Hendrickson, J.R. (1954) Ecology and sistematics of salamanders of the genus Batrachoseps. *Zoology*, **54**, 1-46.

Hewlett, J.D. (1982) Forests and floods in the light of recent investigation. *In: Proceedings of the Canadian Hydrological Symposium*, 1-15 June 1982. pp. 543-560. Fredericton, Canada.

Heyer, A. (1973) Biology of Amphibians. *Journal of Herpetology,* **21**, 41-45.

Heyer, W.R. (1967) A herpetofaunal study of an ecological transect through the Cordillera de Tilaran, Costa Rica. *Copeia*, 259-271.

Heyer, W.R., and Berven K.A. (1973) Species diversity of herpetofaunal samples from similar microhabitats at two tropical sites. *Ecology,* **54**: 642-645.

Heyer, W.R., Donnely M.A., McDiarmid R.W., Hayek L.A.C., and Foster M.S. (1994) Measuring and Monitoring Biological Diversity: Standard Methods for Amphibians. Smithsonian Institution Press, Washington and London.

Hillyard, S.D. 1976. The movement of soil water across the isolated amphibian skin, *Copeia*, **1976**, 314-320.

Hohn, M.E. 1976. Binary coefficient: A theoretical and emperical study. *J. Int. Assoc. Math. Geol*, **8**, 137-150.

Holomuzki, J.R. (1980) Synchronous foraging and dietary overlap of three species of plethodontid salamanders. *Herpetologica*, **36**, 109-115.

Howard, P. (1986) Conservation of tropical forest wildlife in western Uganda. Unpublished annual report to World Wildlife Fund International, Gland, Switzerland.

Huey, R.B., and M. Slatkin (1976) Costs and benefits of amphibian thermoregulation, *Quart. Rev. Biol.*, **51**, 363-384.

Hughes, J.W. and Fahey, T.J. (1994) Litterfall dynamics and ecosystem recovery during forest development. *For. Ecol. Manage*, **63**, 181-198.

Hurlbert, S.H. (1971) The nonconcept of species diversity: a critique and alternative parameters. *Ecology,* **52**, 577-586.

Hurlbert, S.H. (1978) The measurement of niche overlap and some relatives. *Ecology*, **59**, 67-77.

Hutchinson, V.H. (1961) Critical therma maxima in salamanders, *Physiol. Zool.*, **34**, 92-125.

Inger, R.F (1966) The systematics and zoogeography of the Amphibia of Borneo. *Field Diana : Zool.*, **52**. 402 pp.

Inger, R.F. (1969) Organization of communities of frogs along small rainforest streams in Sarawak. *Journal of Animal Ecology*, **80**, 123-148.

Inger, R.F. (1979) Abundances of amphibians and reptiles in tropical forests of Southeast Asia. *In:* Trans. Sixth Aberdeen-Hull Symposium on Malesian Ecology. (A.G. Marshall, Ed). Univ. Hull, Dept. Geography, *Misc. Ser.* no. 22.

Inger, R.F. (1980) Densities of floor-dwelling frogs and lizards in lowland forests of Southeast Asia and Central America. *American Naturalist*, **115**, 761-770.

Inger, R.F. (1989) Observations on several ecological aspects of Bornean anurans in West Kalimantan. Field Report INS/PSDA/II. Indonesian Research Institute, Bogor. Indonesia.

Inger, R.F. and , and Colwell, RK. (1977) Organisation of contiguous communities of amphibians and reptiles in Thailand. *Ecol. Monogr.*, **47**, 229-253.

Inger, R.F., and Greenberg B. (1966). Ecological and competitive relations among three species of frogs (Genus *Rana*). *Ecology*, **47**(5), 746-759.

Inger, R.F., and Marx H. (1961) The food of amphibians, *Exploration du Parc National de l'Upemba. Fasc.,* **64**,1-86.

Inger, R.F., and Stuebing, R.B. (1989) Frogs of Sabah. Sabah Parks Trustees. Sabah, Malaysia.

Inger, R.F., and Taufik A.W. (1996). New Species of Ranid Frogs (Amphibia:Anura) from Central Kalimantan, Borneo. *The Raffles Bulletin of Zoology*, **44**(2), 363-369.

IUCN (1980) *World Conservation Strategy*. IUCN, UNEP, WWF, Morges, Switzerland.

IUCN (1987) *Burma, Forest Conservation Issue*. IUCN, Gland, Switzerland.

Jaeger, R.G. (1970) R.G. Potential exticntion through competition between two species of terrestrial salamanders. *Evolution* , **24**, 632-642.

Jaeger, R.G. (1972) Food as a limited resource in competitionbetween two species of terrestrial salamanders. *Ecology*, **55**, 535-546.

Jaeger, R.G. (1980) Microhabitats of a terrestrial forest salamander. *Copeia*, **1980**, 265-268.

Janzen, D.H. (1974) Tropical black waters rivers, animals, and mast fruiting by Dipterocarpaceae. *Biotropica*, **6**, 69-103.

Johns, A.D. (1983). Selective logging and wildlife conservation in tropical rain forest wildlife: problems and recommendations. *Biol. Cons.* **31**, 355-375.

Johns, A.D. (1986) Effect of habitat disturbance on rain forest wildlife in Brazilian Amazonia. Unpublished report to World Wildlife Fund US, Washington DC, USA.

Johns, A.D. (1988) Economic development and wildlife conservation in Brazilian Amazon. *Ambio*, **17**, 302-306.

Johns, A.D. (1989) Timber, the environment and wildlife in Malaysian rain forests. Final report to Institute of South-east Asian Biology, University of Aberdeen, Aberdeen, Scotland, UK.

Johns, A.D. (1992) Vertebrate responses to selective logging implications for the design of logging systems. *Phil. Trans. R. Soc. Lond.* **B 335**, 437-442.

Jones, K.B. (1988) Distribution and habitat associations of herpetofauna in Arizona: Comparisons by habitat type. pp. 109-128. *In* R. C. Szaro, K.E. Severson, and D.R. Patton (Eds.). Management of Amphibians, Reptiles, and Small Mammals in North America, U.S. Department of Agriculture, Forest Service, General Technical Report RM-166.

Jonkers, W.B.J. (1987) *Vegetation Structure, Logging damage and Silviculture in a Tropical Rain Forest in Suriname.* Agricultural University, Suriname.

Karns, D.R. (1986) Field herpetology: Methods for the study of amphibians and reptiles in Minnesota. James Ford Bell Museum of Natural History, Occasional Paper 18.

Karr, J.R. (1986) Seasonality, resource availability and community diversity in tropical bird communities. *Amer. Nat.*, **110**, 973-994.

Kartawinata K. (1978) Biological changes after logging in lowland dipterocarp forest. *Biotrop Special Publication*, **3**, 27-34.

Kasenene, J.M. (1987). Some observations on the effects of mechanical logging on tropical forests in Papua New Guinea. *Biotrop Spec. Publ*, **3**, 82-86.

Kohn, A.J. (1959) The ecology of Conus in Hawai. *Ecol. Monogr.*, **29**, 47-90.

Kramer, P., Reichenbach, N., Hayslett, M., and Sattler, P. (1993) Population dynamics and conservation of the Peaks of Otter Salamander, *Plethodon hubrichti. J. Herpetol.*, **27**, 431-435.

Krishnamurthy, S.V. (1997) Nyctibatrachus major in Malnad India. *Froglog Newsletter*, January 1997, No.20.

Kurniawan, A. (1987) Notes on the range and ecology of the shrub-dwelling anurans, genus *Philautus. Kukila*, **6,** 71-76.

Labanick, G.M. (1976) Prey availability, consumption and selection in the cricket frog, Acris crepitans (Amphibia, Anura, Hylidae). *J. Herpetol.*, **10**, 293-298.

Lee, A. K. (1968) Water economy of the burrowing frog, *Heleioporus eyrei* (Gray), *Copeia*, **1968**, 741-745.

Lee, J.C. (1980) An ecogeographic analysis of the herpetofauna of the Yukatan Peninsula. *Misc. Publ. Mus. Nat. Hist. Univ. Kansas*, **67**, 1-75.

Leighton, M. and Leighton D.R. (1983) Vertebrate responses to fruiting seasonality within a Bornean rain forest. *In Tropical Rain Forest: Ecology and Management* (S.L. Sutton, T.C. Whitmore and A.C. Chadwick, Eds.). Blackwell Sci. Publ., Oxford, pp. 181-196

Leighton, M., and Wirawan, N. (1984). Catastrophic drought and fire in Borneo tropical rain forest associated with the 1982-1983 El Nino southern oscillation event.. *In: Tropical Rain Forests and the World Atmosphere* (G. Prance, Ed.),. Westview Press, Boulder, Colorado, USA, pp. 75-102.

Lieberman, S.S. (1986) Ecology of the leaf litter herpetofauna of a Neotropical rain forest: La Selva, Costa Rica. *Acta Zoologica Mexicana*, **15**, 1-72.

Liew, T.C. (1978) Forest manipulation and regeneration in Sabah. *Malay. Nat. J.*, **31**, 155-166.

Likens, G.E., Bormann, F.H., Pierce, R.S., and Reiners, W.A. (1978) Recovery of a deforested ecosystem. *Science* (Washington, D.C.), **199**(3), 492-496.

Lindemann, B., and Voute C. (1976) Structure and function of epidermis. *In* R. Linas and W. Precht (Eds), *Frog Neurobilogy, Berlin: Springer-Verlag*, pp. 169-120.

Llyoyd, M., Inger, R.F., and King, F.W. (1968) On the diversity of reptile and amphibian species in a Bornean rain forest. *Amer. Nat.*, **102**, 497-515.

Low, K.S., and K.C. Goh (1972) The water balance of five catchments in Selangor, West malaysia. *J. Trop. Geogr.*, **35**, 60-66.

Lugo, A.E. (1988) Estimating reductions in the diversity of tropical forest species, in *Biodiversity* (E.O. Wilson, and F.M. Peter, Eds.), National Academy Press, Washington DC, pp 58-70.

MacArthur, R.H. (1972) Geographical Ecology. Patterns in the distribution of species. Harper and Row, New York. 269 pp.

MacNamara, M.C. (1977) Food habits of terrestrial adult migrants and immature red efts of the red-spotted newt Notophthalmus viridescens. *Herpetologica*, **33**, 127-132.

Madison, , D.M., and Shoop, C.R. (1970) Homing behavior, orientation, and home range of salamanders. *Science* (Washington, D.C.), **168**, 1484-1487.

Maiorana, A. (1977) Observations of salamanders (Amphibia, Urodela, Plethodontidae) dying in the field. *Journal of Herpetology*, **11**, 1-5.

Maiorana, A.(1977) Observations of salamanders (Amphibia, Urodela, Plethodontidae) dying in the field. *Journal of Herpetology*, **11**, 1-5.

Maiorana, V.C. (1976) Predation, submergent behavior, and tropical diversity. *Evol. Theory*, **1**, 157-177.

Marn, H.M. (1982) The planning and design of the forest harvesting and log transport operation in the mixed dipterocarp forest of Serawak. UNDP/FAO Field Document MAL/76/008, no. 17.

Marn, H.M. and Jonkers W. (1981) Logging Damage in Tropical High Forest. Unpublished report, Forest Department, Serawak.

Marshall, D.B., Chilote, M., and H. Weeks. (1992) Sensitive vertebrates of Oregon.Oregon Department of Fish and Wildlife, U.S.

Marshall, D.B., Chilote, M., and Weeks, H. (1992) Sensitive vertebrates of Oregon.Oregon Department of Fish and Wildlife, U.S.

Martof, B.S. (1953) Territoriality in the green frog, *Rana clamitans*. *Ecology*, **34**, 165-174.

Mathis, A. (1990) Territoriality in a terrestrial salamander: The influence of resource quality and body size. *Behaviour*, **112**, 162-175.

Matsui, T. (1975). The Amphibians in the Oriental Region. Kyoto University Press, Japan.

McComb, W.C., McGarigal, K., and Anthony, R.G. (1993) Small mammal and amphibian abundance in streamside and upslope habitats of mature Douglas-fir stands, western Oregon. *Northwest Sci.*, **67**, 7-15.

McCombe, W.C., McGarigal, K., and Anthony, R.G. (1993) Small mammal and amphibian abundance in streamside and upslope habitats of mature Douglas-fir stands, western Oregon. *Northwest Sci.*, **67**, 7-15.

McElroy, M.B., and Wofsy S.C. (1984) Tropical forests: interactions with the atmosphere. *In Tropical Rain Forests and the World Atmosphere* (G. Prance, Ed. Westview Press, Boulder, Colorado, USA, pp. 33-60.

McNaughton, S.J, and L.L. Wolf (1973) Dominance and the niche in ecological ecosystems. *Science, N.Y.*. **167**, 131-139.

McNeely, J.A., Miller, K.R., Reid, W.V., Mittermeier, R.A. and Werner, T.B. (1990) *Conserving the Worlds's Biodiversity*. IUCN, Gland and WRI/CI/WWF-US/World Bank, Washington. 193 pp.

Mengak, M.T., and Guynn, D.C. Jr. (1987) Pitfalls and snap traps for sampling small mammals and herpetofauna. *American Midland Naturalist*, **118**, 284-288.

Metter, D.E. (1964) A morphological and ecological comparison of two populations of the tailed frog, *Ascaphus truei* Stajneger. *Copeia*, **1964**, 181-195.

Millsap, B.A. and Harrison, W. (1981) Food and foraging habits of common black hawks in Western Arizona. U.S. Dep. Inter., Bur. Land Manage. Phoenix District, A.Z.

Ministry of Forestry (1987) The requirements of environmental impact assessment (EIA) for sustainable forest management of Indonesia. Directorat General of Forest Protection and Nature Conservation and WWF. CTF/INS/039/INS: Forestry Studies. Technical Report No. 2. Jakarta. Indonesia.

Ministry of Forestry (1991) An Agenda for Forestry Sector Development in Indonesia. Directorate General of Forest Utilization, The Indonesian Ministry of Indonesia and FAO. UTF/INS/065/INS: Forestry Studies. Technical Report No. 4. Jakarta. Indonesia.

Ministry of Forestry (1991) An Agenda for Forestry Sector Development in Indonesia. Directorate General of Forest Utilization and FAO. UTF/INS/065/INS: Forestry Studies. *Technical Report*, **4**, 41-44.

Ministry of Forestry (1993) Biodiversity Action Plan for Indonesia. Ministry of National Development Planning/National Development Planning Agency. Jakarta. Indonesia.

Ministry of Forestry (1995) Progress Towards Sustainable Management of tropical Forests (Objective Year 2000). Paper presented in the Nineteenth Session of the International Tropical Timber Council (ITTC), Yokohama, 8-16 November 1995. The Indonesian Ministry of Forestry, Jakarta, Indonesia.

Morin, P.J. (1981) Predatory salamanders reverse the outcome of competition among three species of anuran tadpoles. *Science*, **212**, 1284-1286.

Moyle, P.B. (1973) Effects of introduced bullfrogs, Rana catesbeiana on the native frogs of the San Joaquin Valley, California. *Copeia*, **1973**, 18-22.

Mullen, T. L., and Alvarado, R.H. (1976) Osmotic and ionic regulation in amphibians, *Physiol. Zool.*,**49**, 11-23.

Murphy, M.L. and Hall, J.D. (1981) Varied effects of clearcut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. *Can. J. Fish. Aquat. Sci.*, **38**, 137-145.

Murphy, M.L., Hawkins M.L., and Anderson, N.H. (1981) Effects of canopy modification and accumulated sediment on stream communities. *Trans. Am. Fish. Soc.*, **110**, 469-478.

Myers, C.W., and Rand, A.S. (1969) Checklist of amphibians and reptiles of barro Colorado Island, Panama, with comments on faunal changes and sampling. *Smithsonian Contr. Zool.*, **10**, 1-11.

Myers, N. (1983) *A Wealth of Wild Species*. Westview, Boulder.

Nawangsari, S. (1985) Physiological regulations of Bornean horned frog, *Megophrys nasuta*. Agricultural Bogor Institute Catell Press, Bogor. Indonesia.

Nicholson D.I. (1958) An analysis of logging damage in tropical rain forest in North Borneo. *Malayan Forester*, **21**, 235-45.

Nicholson D.I. (1960) Light requirements of seedlings of five species of Dipterocarpaceae. *Malayan Forester*, **23**, 344-356.

Noel, D.S., Marin, C.W., and Federer, C.A. 1986. Effects of forest clear-cutting in New England on stream macroinvertebrates and peryphyton. *Environ. Manage.*, **10**, 661-670.

Nussbaum, R.A., Brodie, E.D. Jr., and R.M. Storm. (1983) Amphibians and reptiles of the Pacific Northwest. Moscow, ID: University Press of Idaho. 332p.

Orians, G.H. 1975 Diversity, stability and maturity in natural ecosystems. *In* Unifying concepts in ecology. (W.H. van Dobben&R.H. Lowe-McConnell, Eds.). The Hague, The Netherlands: Junk., pp. 139-150.

Packer, P.E. (1967) Forest treatment effects on water quality. *In* International Symposium on Forest Hydrology. Pergamon Press, Oxford, U.K.

Pais, R.C. (1987). The assessment of herpetofaunal species richness by pitfall sampling in hemlock beech in Maine. *Herpetologica*, **50**, 90-97.

Pais, R.C., Bonney, S.S., and McComb, W.C. (1988) Herpetofaunal species richness and habitat associations in an eastern Kentucky forest. Proc. Annu. Conf. Southeast. Assoc. Fish. Wildl. Agencies, **42**, 448-455.

Park, Thomas (1954) Experimental Studies of interspecific competition. Temperature, humidity, and competition in two species of *Tribolium*. *Physiol. Zool.*, **27**, 177-238.

Patric, J.H. (1976) Soil erosion in the eastern foest. *J. For.*, **74**(10), 671-677.

Peachman, J.H.K., and Wilbur, H.M. (1994) Putting declining amphibian populations in perspective: natural fluctuations and human impacts. *Herpetologica*, **50**, 65-84.

Peachman, J.H.K., D.E. Scott, R.D., J.P. Caldwell, L.J. Vitt and J.W. Gibbons (1991) Declining amphibian populations: The problem of separating human impacts from natural fluctuations. *Science*, **253**, 892-895.

Pefaur, J.E., and W.E. Duellman (1980) Community structure in high Andean herpetofauna. *Trans. Kansas Acad. Sci.*, 83, 45-65.

Petranka, J.W. (1994) Response to impact of timber harvesting on salamanders. *Conserv. Biol.* **8**, 302-304.

Petranka, J.W., Brannon, M.P., Hopey, M.E., and Smith, C.K. (1994) Effects of timber harvesting on low elevation populations of southern Appalachian salamanders. *For. Ecol. Manage*, **67**, 135-147.

Petranka, J.W., Eldridge M.E., and Haley, K.E. (1993) Effects of timber harvesting on southern Appalachian salamanders. *Conservation Biology*,**7**, 363-370.

Phelps, J.P. and Lancia, R.A. (1995) Effects of clearcut on the herpetofauna of a South Carolina Wwyman Forest. *Brymleyana*, **22**, 31-45.

PHT Kayu Mas. 1994. Evaluation on vegetation damages caused by selective logging in Kayu Mas Forestry Logging Concession, Central Kalimantan. Camp Tumbang Manyak. Sangai. Central Kalimantan, Indonesia.

Pianka, E.R. (1977) "Reptilian species diversity," . *In* C. Gans and D.W. Tnkle (Eds.) Biology of Reptilia, Vol. 7, New York: Academic Press, pp 1-34.

Pimm, S.L. (1984) Complexity and stability: another look at MacArthur's original hypothesis. *Oikos*, **33**, 351-357.

Plumptre, R.A. and Earl D.E. (1984) Integrating small industries with management of tropical forest for improved utilization and higher future productivity. Paper presented to IUFRO Division P5.01 Meeting on Properties and Utilization of Tropical Timbers, Manaus, Brazil.

Potter, G.L., H.W. Ellsaeser, M.C. McCracken and F.M. Luther (1975). Possible climatic impact of tropical deforestation. *Nature*, **258**, 697.

Pough, F.H, Smith, E.M., Rhodes, D.H., and Collazo, A. (1987) The abundance of salamanders in forest stands with different histories of disturbance. *Forest Ecology and Managemen*t, **20**, 1-9.

Pough, F.H., Taigen, T.L., Stewart, M.M., and Brussard, P.F. (1983) Behavioral modification of evaporative water loss by a Puerto Rican frog. Ecology, **64**, 244-252.

Pough. F.H. (1980) The advantages of ectothermy for tetrapods. *Am. Nat.*, **115**, 92-112.

Pough. F.H. (1982) Ecological correlates of terrestrial amphibians exercise physiology. *Oecologia*, **50**, 40-45.

Pounds, J.A., and Crump, M.L. (1994) Amphibian declines and climate disturbance:"the case of the golden toad and the harlequin frog". *Conservation Biology* , **8**(1), 72-85.

Powers, V.N., and Tietjen, W.L. (1974) The comparative food habits of sympatric and allopatric salamanders, *Plethodon glutinosus* and *P jordani* in eastern Tennessee and adjacent areas. *Herpetologica*, **30**, 167-175.

Poynton, J.C., and and Pritchard S. (1976). Notes on the biology of *Breviceps* (Anura: Microhylidae). *Zool. Africana*, **11**, 313-318.

Proctor, J. (1995). A short review on the UK-Indonesia Tropical Forest Management Project at Camp 48, Central Kalimantan. Interim Report to the Edinburgh Centre for Tropical Forests (ECTF). Jakarta. Indonesia.

Proud, K. (1995) Forestry Research (Quartely Report). Indonesia-UK Tropical Forest Management Project, July-September, 1995. Central Kalimantan. Indonesia.

Raich, J.W. (1989) Seasonal and spatial variation in the light environment in a tropical dipterocarp forest and gaps. *Biotropica*, **21**, 299-302.

Raphael, M.G. (1988) Long-term trends in abundance of amphibians, reptiles, and mammals in Douglas-fir forests of northwestern California. *In* Szaro, R.C., Severson R.C., Patton, K.E., D.R., Eds.). Management of amphibians, reptiles, and small mammals in North America. Gen. Tech. Rep. RM-166. Ft. Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Range and Experiment station, pp. 23-31.

Raphael, M.G. (1991) Vertebrate species richness within and among seral stages of Douglas-fir/hardwood forest in north-western California. *In*: Wildlife and vegetation of unmanaged Douglas-fir forests. *E,* (L.F. Ruggriero, K.B. Aubry, A.B. Carey, and M.H. Huff, Eds.). USDA For. Serv. Gen. Tech. Rep. PNW-GTR-285. pp. 415-423.

Raxworthy, C.J (1988) Reptiles, rain forest and conservation in Madagascar. *Biol. Cons.*, **43**, 181-211.

Ray, C. (1958) Vital limits and rates of desiccation in salamanders. *Ecology*, **39**, 75-83.

Raymond, L.R., and Handy, L.M. 1991. Effects of clearcut on a population of the mole salamander, *Ambystoma talpoideum*, in adjacent unaltered forest. *J. Herpetol.* **25**: 509-512.

Redwood , J.F. (1960) An analysis of logging damage in lowland rain forest. *Nigerian Forestry Information Bulletin* (New Series), **10**, 5-16.

Reid, L.M., and Dunne T. (1984) Sediment production from forest road surfaces. *Water Resour. Res.*, **20**, 1753-1761.

Rice, R.M., Tilley, F.B., and Datzman, P.A. (1979) A watershed' response to logging and roads: South Fork of Caspar creek, California, 1967-1976. *USDA For. Serv. Res. Pap.* PSW-146, 12 pp.

Roughgarden, J. (1983) Competition and theory in community ecology. *Amer. Nat.*, **122**, 583-601.

Rudolph, D.C., and Dickson, J.G. (1990) Streamside zone width and amphibian and reptile abundance. *Southwest. Nat.*, **35**, 472-476.

Ruggiero, L.F., Aubry K.B., Carey A.B. and Huff, M.H. (1991) Wildlife and vegetation of unmanaged Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-285. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 533p.

Ruibal, R., L. Tevis, Jr., and V. Roig (1969) The terrestrial ecology of the spadefoot toad *Scaphiopus hammondii*, *Copeia*, **1969**, 571-584.

Sayer, J.A. and Whitmore T.C. (1991) Tropical moist forests: destruction and species extinction. *Biological Conservation*, **55**, 199-214.

Schmitt, A. 1990. Effects of selective logging on vegetation structure and composition of Bornean rain forest in Kayu Mas Logging areas.

Schoener, T.W. (1983) Field experiments on interspecific competition. *Amer. Nat.*, **122**, 240-285.

Schultz, J.P. (1960). Ecological studies on rain forest in northern Suriname. Noord-Hollandsche Uitgevers Maatschappij, Amsterdam, The Netherlands.

Scivener, J.C., and Andersen, B.C. (1984). Logging impacts and some mechanisms that determine the size of spring and summer populations of stream dwelling communities in Carnation Creek , British Columbia. *Can. J. Fish. Aquat. Sci.*, **41**, 1097-1105.

Scott, N.J. Jr. (1976). The Abundance and Diversity of the Herpetofaunas of Tropical Forest litter. *Biotropica* **8**(1), 41-58.

Scott, N.J. Jr. (1982). The herpetofauna of forest litter plots from Cameroon, Africa. *U.S. Fish and Wildl. Res. Rept.*, **13**, 145-150.

Scott, N.J., Andrew, A., and Steve, J. (1983) Species listing of the herpetofauna in Tropical forest litter. *Biotropica*, **8** (1), 41-45.

Sedell, J.R., and Swanson, F.J. (1984) Ecological characteristics of streams in old-growth forests of the Pacific Northwest. *In* W.R. Meehan, T.R. Merrell, Jr. and T.A. Hanley (Eds.), Fish and Wildlife Relationships in Old-Growth Forests. Proc. Symp. 12-15 April 1982, at Juneau, AK. American Institute of Fishery Research Biologists, pp. 33-46.

Sexton, M. (1988) A note of single-species population of aquatic amphibians in Peninsular Malaysia. *Herpetologica*, **34**, 409-416.

Simberloff, D. (1983) "Sizes of coexisting species," . *In* D.J. Futuym and M. Slatkin (Eds.). Coevolution, Sunderland, Massachusetts: Sinauer Assoc., pp. 404-430.

Simberloff, D. (1986) Are we on the verge of mass extinction in tropical rain forests?, in *Dynamics of Extinction* (D.K. Elliott, Ed.), Wiley, New York.

Simon, J.L. (1986) Disappearing species, deforestation and data. *New Scientist*, May 15, 1986.

Sinch, U. (1990) Migrationand orientation in anuran amphibians. *Ethol. Ecol. Evol.* **2**, 65-79.

Sites, J.W. Jr. (1978) The foraging strategy of the dusky salamander, *Desmognathus fuscus* (Amphibia, Urodela, Plethodontidae): an empirical approach to predation theory. *J. Herpetol.*, **12**, 373-383.

Skorupa, J.P. 1986. Responses of rain forest primates to selective logging in Kibale forest, Uganda: a summary report. *In* K. Bernirschke (ed.). Primates: the road to self -sustaining populations,. Springer-Verlag, New York, pp. 55-70.

Smith, M.A. (1930) The Reptilia and Amphibia of the Malay Peninsula. *Bull. Raffless Mus.*, **3**, 149 pp.

Sokal, R.R., and Michener, C.D. (1958) A statistical method for evaluating systematic relationships. University of Kansas Science Bulletin, **38**, 1409-1438.

Sokal, R.R., and Rohlf, F.J. 1995. *Biometry*: The principles and practice of statistics in biological research. W.H. Freeman and Company. New York. 887 pp.

Soong, N.K., Haridas, G., Yeoh, C.S., and Tan P.H. (1980) Soil erosion and conservation in Peninsular malaysia. Rubber Research Institute of Malaysia, Kuala Lumpur, Malaysia.

Sorensen, T. 1948. A method of eatablishing groups of equal amplitude in plant sociology based on similarity of species content, and its application to analyses of the vegetation on Danish commons. *Det Kongelige Danske Videnskkabernes Selskab, Biologiske Skrifter*, **5**, 1-34.

Sormin B. (1992). Conserving Biological Diversity in Managed Tropical Forests. *Kalawarta* **4**, 45-50.

Soule', M.E. 1986. Reconstructed dynamics of rapid extinction of tropical forests. *Biology International Special issue* **22**.

Spotila, J.R. (1972) Role of temperature and water in the ecology of lungless salamanders. *Ecol. Monog.*, **42**, 95-125.

Stebbins, R.C., and Cohen, N.W. (1995) A natural history of amphibians. Princeton University Press, Princeton. N.J.

Steven, A.E., and Bruce, R.C. (1988) Ecological genetics of the salamander Desmognathus quadramaculatus from disturbed watersheds in the Southern Appalachian Biosphere Reserve Cluster. *Conserv. Biol.*, **2**, 194-205.

Stewart, G.H., Rose, A.B., and Veblen T.T. (1991) Forest development in canopy gaps in old-growth beech (Nothofagus) Forests, New Zealand. *J. Veg. Sci.*, **2**, 679-690.

Strong, D.R., Syska, L.A., and Simberloff, D.I. (1979) Tests of community-wide character displacement against null hypotheses. *Evolution*, **33**, 897-913.

Struhsaker, T.T. (1987) Forestry issue and conservation in Uganda. *Biological Conservation*, **39**, 209-34.

Stuebing, R. (1992) Notes on quantitative analysis in community ecology. Unpublished Report. Sabah Park. Malaysia.

Swanson, F.J., and Lienkaemper, G.W. (1978) Physical consequences of large organic debris in Pacific Northwest streams. USDA For. Serv. Gen. Tech. Rep. PNW-69, 12 pp.

Swanson, F.J., Frankel, J.F, and Sedell, J.R. (1990) Landscape patterns, disturbance, and management in the Pacific Northwest, USA. *In* Changing landscape: an ecological perpective. (I.S. Zonnevald and R.T.T. Formann, Eds.). Springer-Verlag, New York, pp. 191-213.

Szaro, R.C. (1988) The management of amphibians, reptiles, and small mammals in North America: Historical perspective and objectives. *In* Management of amphibians, reptiles, and mammals in North America. (R.C. Szaro, K.E. Severson, and D.R. Patton, Eds.) USDA For. Serv. Gen. Tech. Rep. RM-166. pp. 1-4.

Thiollay, J.M. (1992). Influence of Selective Logging on Bird Species Diversity in a Guianan Rain Forest. *Conservation Biology*, **6** (1), 47-63.

Thorson, T.B. 1964. The partitioning of body water in Amphibia. *Physiol. Zool.*, **37**, 395-399.

Tirta, M (1990). A synopsis of the ecology and behaviour of Brown Tree Toad, *Pedostibes hosei*. Survey Report to LIPI, Vol. VIII (2). Bogor. Indonesia.

Toft, C.A. (1980) Feeding Ecology of thirteen species of anurans in a seasonal tropical environment. *Oecologia*, **45**, 131-141.

Trueb, L. (1971) Phylogenetic relationships of certain Neotropical toads with the description of a new genus (Anura:Bufonidae) and Australian myobatrachids. *Contrib. Sci. Los Angeles Co. Mus. Nat. Hist.*, **216**, 1-40.

Turner, F.B. (1959) An analysis of the feeding habits of Rana pretiosa in Yellowstone Park, Wyoming. *Amer. Mildl. Nat.*, **61**, 403-413.

Tyler, M.J. (1976) Frogs. Sydney: Collins, Ltd.

Tyler, M.J. (1991). Declining amphibian populations-a global phenomenon: An Australian perspective. *Alytes*, **9**, 43-50.

Uhl, C. and Viera, I.C.G. (1989). Ecological impacts of selective logging in the Brazilian Amazon: a case study from the Paragominas region of the state of Para. *Biotropica*, **21**, 98-106.

Uhl, C., and Buschbacher, R. (1985) A disturbing synergism between cattle ranch burning practices and selective tree harvesting in the eastern Amazon. *Biotropica*, **17**, 265-268.

Van Horne, B. 1983. Density as a misleading indicator of habitat quality. *J. Wildl. Manage.*, **47**:893-901.

Vial, J.L., and Saylor, L. (1993) The status of amphibian populations. Working document No. 1. Declining Amphibian Populations Task Force, World Conservation Union, Gland, Switzerland.

Vitt, L.J., Cladwell, J.P., Wilbur, H.M., and Smith, D.C. (1990) Amphibians as harbingers of decay. *Bioscience*, **40**, 418.

Wake, D.B. (1991) Declining amphibian populations. *Science* (Washington, D.C.), **253**: 860

Walls, S.C., Blaustein, A., and Beatty J.J. (1992) Amphibian biodiversity of the Pacific Northwest with special reference to old-growth stands. *Northwest Environmental Journal.*, **8**, 53-69.

Welsh, H.H., Jr. and Lind, A.J. (1991) The structure of the herpetofaunal assemblage in Douglas-fir/hardwood forests of northwestern California and Southeastern Oregon. *In* Ruggiero, Leonard F.; Aubry, K.B.; Carey, A.B., and M.H. Huff, Eds.). Wildlife vegetation of unmanaged Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-285. Portland, OR: U.S. Department of, Forest Service, Pacific Northwest Research Station, pp. 394-413.

Welsh, H.H.Jr. (1990) Relictual amphibians and old-growth forests. *Alytes*, **9**, 33-42.

White, J.T. Lee (1992) Vegetation history and logging disturbance: Effects on rain forest mammals (with special emphasis on elephants and apes) in the Lope reserve, Gabon, Africa. PhD, Thesis, University of Edinburgh. UK.

White, L.D., Harris, L.D., Johnson, J.E., and D.G. Milchunas. 1975. Impact of site preparation on flatwood wildlife habitat. Proc. 29th. Ann. Conf., SE Assoc. Game Fish Comm., pp. 347-353.

White, R.L. (1977). Prey selection by the rough skinned newt (Taricha granulosa) in two pond types. *Northwest Sci.*, **51**: 114-118.

Whitear, M. (1977) A functional comparison between the epidermis of fish and amphibians. *Symp. Zool. Sc. London*, **39**, 291-313.

Whitmore, T.C and Sayer, J.A. (1992) *Tropical Deforestation and Species Extinction.* Chapman & Hall, London.

Whitmore, T.C. (1984) *Tropical Rain Forests of the Far East*, 2nd edn, Clarendon Press, Oxford.

Whitmore, T.C. (1990) *An Introduction to Tropical Rain Forests*. Clarendon Pess, Oxford.

Whitmore, T.C. and Silva, J.N.M. (1990) Brazil rain forest timbers are mostly very dense, *Commonwealth Forestry Review*, **69**, 87-90.

Whitten, A.J., Damanik, S.J., Anwar, J. And Hisyam, N. (1984) *The ecology of Sumatra*. Gadjah Mada University Press, Yogyakarta.

Wiest, J.A. Jr. (1982) Anuran succession at temporary ponds in a post oak-savanna region of Texas. *U.S.Fish and Wildlife Res. Rept.*, **13**, 39-47.

Wilbur, H.M. (1980) Complex life cycles. *Annu. Rev. Ecol. Syst*, **11**, 67-93.

Wilson C.C. and W.L. Wilson (1975). The influence of selective logging on primates and some other vertebrates in East Kalimantan. *Folia Primatol*, **23**, 245-274.

Wilson, E.O. (1988) The current state of biological diversity, in *Biodiversity* (eds E.O. Wilson and F.M. Peter), National Academy Press, Washington, DC, pp. 3-18.

Woinarski, J.C.Z., and Gambold N. (1987). Gradiennt Analysis of Herpetofauna in Stage III of Kakadu National Park, Australia. *J. Austr.*, **4**, 42-45.

Wolf, E.C. (1987) On the brink of extinction: conserving the diversity of life. *World Watch Paper*, **78**.

World Resource Institute (1990) *World Resource* 1990-91. Oxford University Press, Oxford.

Wyman, R.L. (1988) Soil acidity and moisture and the distribution of amphibians in five forests of southcentral New York. *Copeia*, **2**, 394-399.

Wyman, R.L. (1990) What's happening to the amphibians? *Conserv. Biol.*, **4**, 350-352.

Wyman, R.L., and Jancola, J. (1992) Degree and scale of terrestrial acidification and amphibian community structure. *J. Herpetol.*, **26**(4), 392-401.

Yager, C. (1984) Primate population density and food availability in Natal Lengkuas, Central Kalimantan. Interim report to Tanjung Putting national Park, Palangkaraya. Indonesia.

Zimmerman, B.L. and Rodrigues, M.T. (1990) Frogs, snakes, and lizards of the NPA-WWF reserves near Manaus, Brazil. *In* Four Neotropical Rainforests, (A.H. Gentry, Ed.). New Haven, Connecticut: Yale University Press, pp. 426-54.

Zug, G.R. (1979) Distribution and ecology of amphibians in the tropics. *J.Morph.*, **164**, 41-54

Zug, G.R. (1993) Herpetology: an introductory biology of amphibians and reptiles. Academic Press. Inc., San Diego, Calif.